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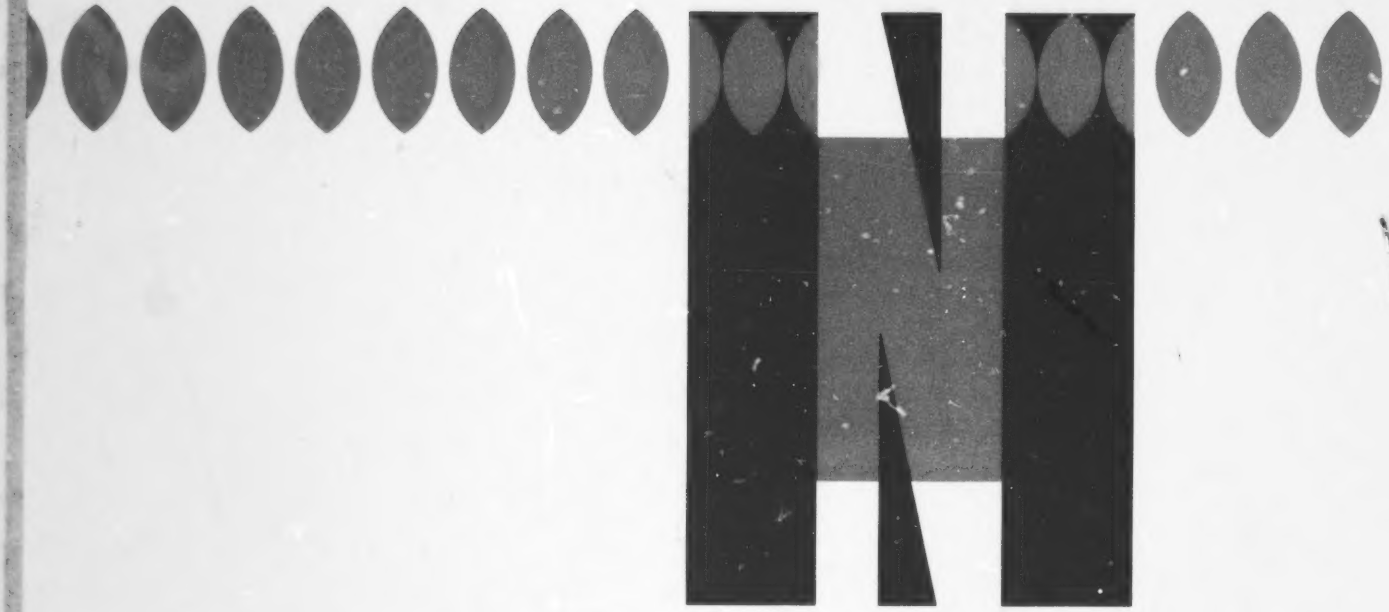

MAY 15-18, 1978
Sacramento, California

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The purpose of this conference was to review the comprehensive national research effort currently being performed in the field of nitrogen management in irrigated agriculture. The National Science Foundation, the U.S. Environmental Protection Agency, and the University of California's Department of Soil and Environmental Sciences jointly sponsored the conference. The comprehensive research approach has proven successful in harmonizing conflicting projects where economy of production and the protection of environmental quality have come into opposition. The growth of public concern over nitrate pollution of water supplies has resulted in the government's increased encouragement, through Federal grants, of the agricultural and environmental aspects of soil nitrate research. The diverse research topics presented at the conference included nitrogen's role in agricultural production, its relation to water quality, crop utilization, removal by irrigated crops, nitrate leaching from soils, economic impacts and methodology, case studies, mathematical studies, and diagnostic identification techniques for nitrogen fertilization.

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MANAGEMENT OF NITROGEN IN IRRIGATED AGRICULTURE

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PREFACE

Traditionally, the main interests in management of nitrogen in irrigated agriculture have been related to agronomic and economic matters. Passage of the Federal Water Pollution Control Act in 1965 broadened the scope of interest in nitrogen management substantially by introducing environmental concerns. It soon became apparent, however, that there were some important gaps in our knowledge that would have to be closed in order to deal effectively with these expanded interests. If nitrogen were to be managed in a manner that would meet both the traditional needs of growers and the environmental concerns of the general public, new information would have to be developed by research. Specifically, the extent of nitrogen losses from irrigated lands and the mechanisms involved were not adequately understood to develop best management practices.

In the intervening years, numerous grants for research on these matters were made by the U.S. Environmental Protection Agency's Office of Research and Development, and by the National Science Foundation to universities, federal government agencies, and private research groups. Linked with these funds was the understanding that the research programs would include a technology transfer activity for the benefit of potential users. Results of the research have been published in various journals and have been presented at meetings with local or regional focus.

These Proceedings represent a National Conference on Nitrogen Management in Irrigated Lands, designed to bring together users representing a diversity of interests throughout the country for an integrated review of the findings from the comprehensive national research effort. The accumulated information needs to be utilized as fully as possible by the many individuals and organizations making decisions in current water quality planning. This will help ensure the emergence of appropriate and viable solutions to our water quality problems -- solutions that are sensitive to both local and national needs and priorities.

The program planning committee for the Conference consisted of the following:

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The financial support of the National Science Foundation through Grant No. ENV76-10283 A01 and the U.S. Environmental Protection Agency through Grant No. R805394-1, is hereby gratefully acknowledged. Without this support the Conference could not have been held and these proceedings could not have been produced. We are indebted to each author for submitting manuscripts and for making presentations during the Conference. We appreciate the help of Winnie Gillette and Art Laag in the preparation of these proceedings.

P. F. Pratt
R. L. Branson

April 1978
University of California
Riverside, CA 92521

RESOURCE MANAGEMENT AS A MEANS OF POLLUTION CONTROL IN IRRIGATED AGRICULTURE

The development of effective pollution control strategies requires an adequate understanding of the factors which affect the pollutant parameters in the systems we desire to protect. The control of nitrogen as a pollutant in the environment is a matter of concern since it is an essential element of life processes and as such the consequences of any control strategy should be carefully examined. The nitrogen cycle is one of the principle bio-chemical processes which supports animal and plant life on the "spaceship earth" on which we reside. Left to the elements of nature a balance has developed which provides continuity of life in many forms.

Through modern technology and understanding of science, mankind has exerted a degree of control over the natural system to benefit the perceived needs of man. His ability to provide foodstuffs in excess of individual needs is a prime example of how he has learned to manipulate the natural system. However, inherent in this ability to control the system is the responsibility to manage it properly both for presently perceived needs and for needs of future generations. This is the challenge we face -- what actions should we take and what are the consequences if we act or do not act.

Obviously, the answers are not easy to obtain since there are conflicting needs to resolve. Yet decisions will be made which will affect all of society. A prudent approach would be to acquire a working knowledge of the system we are trying to manage; develop an understanding of how management options affect the system; and formulate alternative management strategies which allow us to objectively weigh the impact of action or inaction.

This conference was conceived to provide you with an opportunity to better understand the role of nitrogen in irrigated agriculture. The Environmental Protection Agency, chartered to enhance the quality of the environment, is actively seeking an understanding of what can be controlled and what should be controlled to provide for the nation's present and future needs. Through coordination of research efforts among federal and state agencies, attention can be focused on specific problems to develop appropriate solutions. This Conference is one such endeavor to provide not only research results but also opportunity to utilize the researchers as resource persons by the various publics represented here at the Conference.

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NATIONAL INTEREST IN NITRATES IN WATERS

When the National Science Foundation organized its program on Research Applied to National Needs (RANN) in 1971, it sought to encourage research in areas of science having application in more than one of the Federal mission agencies. It was felt that research funds could be used efficiently by having a common source of funding for investigations spanning the interests of multiple agencies. Moreover, it was recognized that agency missions can sometimes clash, as in the cases where economy of production and protection of environmental quality come into conflict. These are situations in which a comprehensive research approach can be useful, an approach that seeks to harmonize, as well as possible, society's need for the production of goods and the production of a clean and healthy environment. Yet the needed comprehensive approach is ever in danger of falling apart if it has to depend upon the coordination of isolated projects supported by different agencies, each working under its own legislative mandate.

Early in its program in support of applied research, the National Science Foundation was aware of public concern about nitrate pollution of water supplies. The Foundation therefore encouraged the development of proposals to combine the agricultural and environmental aspects of research on soil nitrate in a unified approach designed to harmonize the objectives of economic crop production and environmental protection. Successful initiatives from Washington University in St. Louis and from the University of California were the response to the Foundations' interest in this area. The Washington University project, recently completed, dealt with conditions in the midwestern cornbelt.

The California project, under the leadership of Dr. Parker F. Pratt has involved the Berkeley, Davis and Riverside campuses of the university. The results of this six-year program now in its final stage, provide much of the input to this conference on nitrate in the irrigated lands of the West. The National Science Foundation is pleased to join with the Environmental Protection Agency in sponsoring this Conference to assist in the utilization of the research it has supported along with other related investigations to be reported here. It is the hope of the Foundation that its support of work in this area will have helped to show how an integrated research effort can help to find acceptable compromises between environmental protection and the use of nitrogen fertilizers for economic crop production.

Richard A. Carrigan
National Science Foundation
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THE INDISPENSABLE ROLE OF NITROGEN IN AGRICULTURAL PRODUCTION

R. A. Olson[1/]

ABSTRACT

The indispensable role of nitrogen in food and fiber production for the world's people cannot be disputed. There is no substitute for nitrogen in its essential roles as a component of the chlorophyll and protein constituents of crop plants. The quantity required for obtaining an economic yield of most crops exceeds that of all other soil-derived essential nutrients. The advent of relatively cheap fertilizer nitrogen in the 1950's caused radical increases in yields obtainable with most crops in the developed countries and provided the spark that ignited the Green Revolution in many of the Less Developed Countries in the 1960's as well. Its preeminence in the food production chain notwithstanding, nitrogen has been subjected recently to more critical surveillance than any other element in agriculture by reason of energy expended in its conversion into fertilizers, its monetary cost to the farmer, and its potential role as environmental pollutant. The economic and environmental problems can be minimized, however, by matching rate and timing of applied nitrogen with the amount likely to be provided by the soil during the growing season and with the water regime afforded. The agricultural sector must achieve this matching objective promptly if it is not to be condemned by the rest of society in the long term.

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NATURE AND OCCURRENCE

Nitrogen is one of the most plentiful of all the elements known to be essential for plant growth. It constitutes 78% of the atmosphere as elemental nitrogen (N_2) gas and is present in substantial quantities combined with other elements in the sedimentary rocks of the earth's crust and in the organic fraction of the soil. The latter constitutes the major natural reservoir of nitrogen likely to become available for crops over the long term. A representative figure for the surface soil in a native stand of grass or forest, never cultivated, might be in the order of 3% organic matter of which approximately 5% would be nitrogen. Of course there are soils developed in old lake beds that can range up to nearly 100% organic matter (peat), and on the other extreme desert soils that contain barely perceptible amounts. By knowing the weight of the soil, the few thousand kg nitrogen/ha-30 cm is readily calculated. Through microbial activity from 2 to 4% of this total is released as mineral ammonium and nitrate available to plants over the short time interval of a growing season.

So the growing plant may be likened to the ship-wrecked sailor with oceans of water about but nary a drop to drink. There are more than 67,000 metric tons of nitrogen/ha in the atmosphere above and even larger amounts in the rocks below, none of which is useable by the plant due to the inert elemental state of that above and the inaccessibility of that below. In essence, there is a larger total amount present in the biosphere but with proportionately less in fixed forms available for immediate crop production relative to crop needs than exists with any other essential plant nutrient. Nitrogen contained in the grain of most commonly grown crops exceeds the combined total of all other soil-derived essential nutrients with the exception of carbon, hydrogen and oxygen. As a consequence nitrogen is the most extensively deficient element for optimum crop production world-wide (Donald et al, 1963) and greatly exceeds all others in tonnage applied as fertilizer today.

ROLE OF NITROGEN IN PLANT GROWTH

There is no substitute for nitrogen in the essential roles it performs in plant growth. The all-important function of energy intake from solar radiation effected by chlorophyll could not be served were it not for the nitrogen component which completes that remarkable photoreceptor molecule (Figure 1). Nitrogen is also one of the necessary constituents of amino acids, the building blocks of protein so essential to the nutrition of animals and man. The proteins formed in the plant afford functional roles in the plant's growth as well, many of which are enzymes controlling metabolic processes including the synthesis of protein. In the case of cereals, threshold levels of protein exist below which grain does not form.

Adequacy of nitrogen in plant cells is needed for the utilization of carbohydrates produced by photosynthesis. Under conditions of deficiency, excess carbohydrate deposition occurs in vegetative cells with resulting cell thickening, less protoplasm formation and ultimately reduced plant succulence. Sufficiency of nitrogen to the plant is mandatory not only for luxuriant top growth but also for the development of an extensive and vigorous root system. Thus, Holt and Fisher (1960) measured an approximate 20% increase in the mass of roots under nitrogen fertilized Coastal Bermuda grass as compared to unfertilized grass, and Kmoch et al (1957) reported notably deeper root development from applied nitrogen for extraction of existing deep subsoil moisture by winter wheat as did Olson et al (1964b) for corn and grain sorghum.

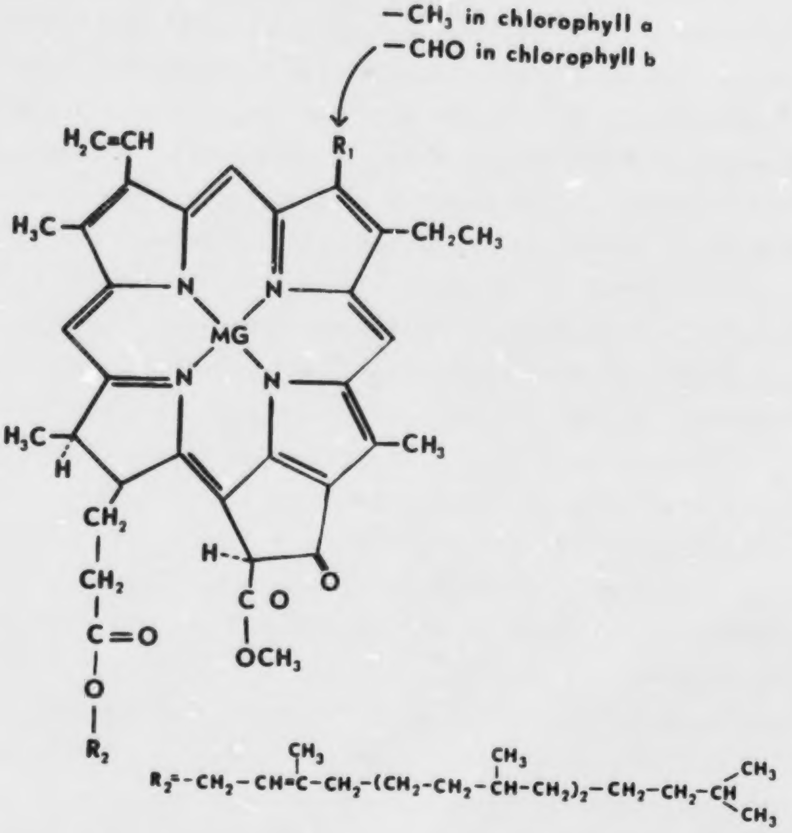
−CH₃ in chlorophyll a
−CHO in chlorophyll b
R₁
H₂C=CH
H₃C
−CH₂CH₃
N
MG
CH₃
H
CH₂
C=O
O
R₂
C O
OCH₃
R₂=-CH₂-CH=C-CH₂-(CH₂-CH₂-CH-CH₂)₂-CH₂-CH₂-CH

Figure 1. Structural formula of the chlorophyll molecule.

CROP REQUIREMENTS

Figure 2 summarizes approximate quantities of nitrogen and other major nutrients present in good yields of five major agricultural crops. These cannot be taken as precise values for the several elements since nutrient composition of crops varies substantially with differences in soil nutrient availability, plant genotype and environmental conditions from one place to another. It will be noted first in these approximations that the amount of nitrogen removed/crop generally exceeds that of all other nutrients, especially in the case of grain crops where the forage is returned to the soil. The second observation of significance is the 110 to 170 kg nitrogen/ha removed by the harvested portion of the crop, excepting the legumes (in this case soybeans) which contain much greater amounts but which fix much of the nitrogen on their own. With annual removals of this magnitude by non-legumes it is evident that total depletion of the native nitrogen in the surface 30 cm of a medium textured virgin soil of 3% organic matter content would be accomplished in an approximate 50-year period if there were no replenishment.

Not only is the total quantity of nitrogen released during a growing season important, but the rate of release assumes prime significance during the crop's grand period of growth. The corn crop producing some 10,000 kg/ha and utilizing around 170 kg nitrogen/ha absorbs almost 3% of the amount/day during the two weeks just before silking, or about 4 kg nitrogen/ha daily. Similar amounts are utilized by wheat during the maximum vegetative stage and somewhat more by soybeans (Figure 3 and Hanway, 1962). Accordingly, the crop manager must assure that nitrogen is plentifully available throughout this period of high demand without affording excessive quantities that might ultimately contribute to environmental pollution.

Much more will be presented on nitrogen removal by irrigated crops in a subsequent paper.

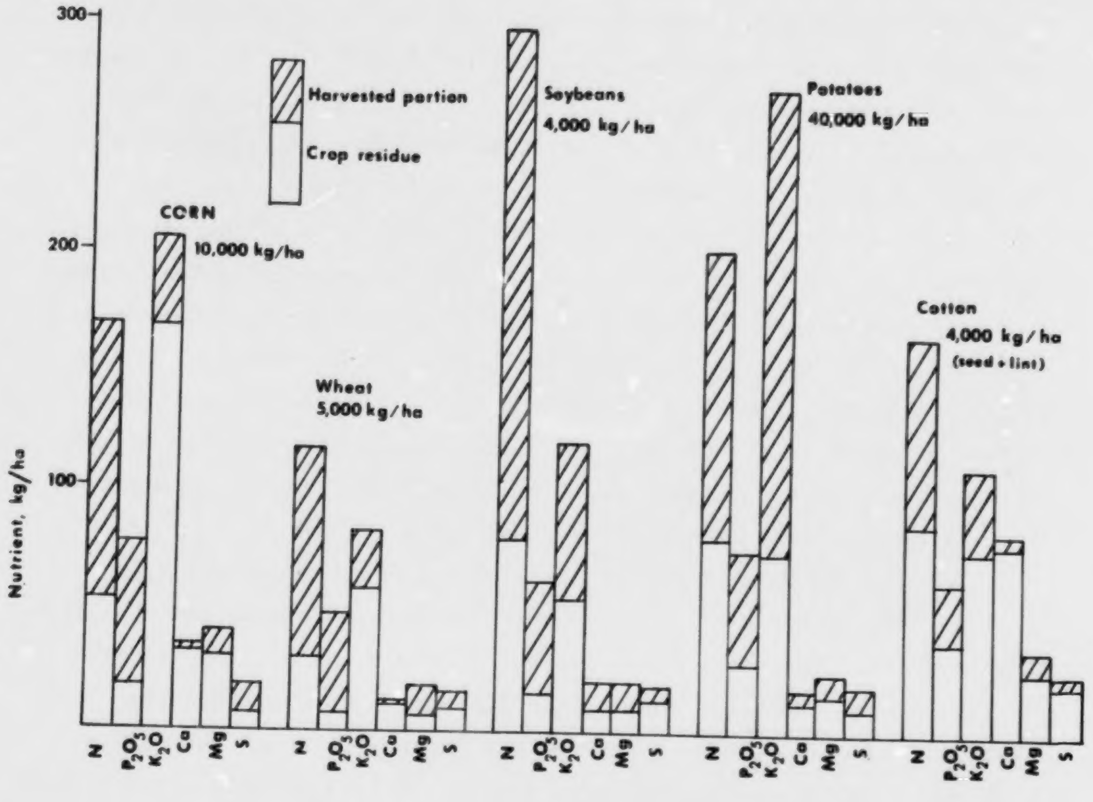

Figure 2. Nutrients contained in harvested portion and residues of good yields of major agricultural crops.

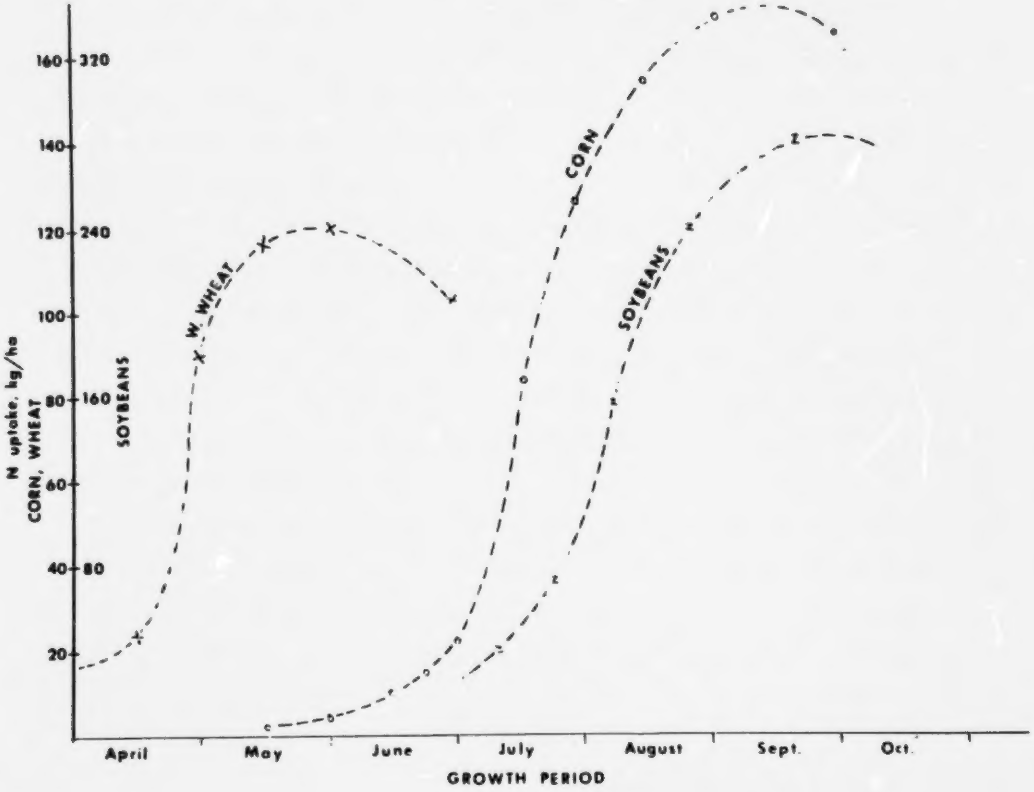

Figure 3. Rate of uptake of nitrogen by winter wheat, corn and soybeans under Nebraska environmental conditions.

NITROGEN SUPPLYING MEASURES FOR CROP PRODUCTION

Organic Methods

Supplemental nitrogen for benefitting crop growth has been achieved through the centuries of time by the growth of a leguminous crop and by the addition of animal manures and human sewage to the soil. During the Golden Age of Greece, several hundred years B.C., authors wrote of the great value of manures for preserving the productivity of the land, and in ancient Rome the relative value of several leguminous plants for soil improvement was generally known.

Well into the 20th century farmers world-wide depended primarily on manures and legumes in rotation for supplying the nitrogen requirements of crops after high levels of native soil fertility had been depleted. Beginning with the chemical age of agriculture in the years immediately following World War II, however, greatly modified traditional concepts of soil fertility maintenance developed in the industrialized countries. During the earlier organic age, as now, there was never more than about 15 to 20% of the nitrogen required for crop production available from the animal manure source (CAST, 1977) while the leguminous green manure crop occupies the land for a year or more when neither food nor fiber crop could be grown. Also the legume's heavy water requirement precluded its use in the drier climatic regions.

Advent of Fertilizer Nitrogen

Technological advances hastened by the urgencies of war resulted in facilities for fixed nitrogen synthesis which were released for peaceful uses at the end of World War II. A burgeoning world population accompanying enhanced health protection measures and the recognition of expanding world food requirements prompted a tremendous growth in fertilizer nitrogen manufacturing facilities. With the resulting substitution of fertilizer nitrogen for the old traditional nitrogen sources, fertilizer nitrogen consumption soon came to exceed the tonnages of other major nutrients (Table 1).

Table 1. Consumption of fertilizer nitrogen, phosphate and potash in the world, 1938-76.

Year	Consumption of N	Consumption of P_2O_5	Consumption of K_2O
	million mt of nutrient*		
1938-39	2.6	3.6	2.8
1953-54	5.2	6.3	5.7
1959-60	9.7	9.7	8.6
1964-65	16.8	13.8	12.1
1967-68	24.2	16.7	14.1
1973-74	38.7	24.2	20.7
1975-76	40.9	26.2	24.7

* mt = metric tons

Note that nitrogen consumption has increased over 15 times in the 37 year period up to 1976 while use of the two other major nutrients increased about 8 times, such that ratio of nitrogen-phosphate-potash expressed as N-P_2O_5-K_2O changed from .29-.40-.31 to .44-.30-.26. Approximately one-fourth of the total nitrogen used in 1975-76 was consumed in the U.S. The crop itself cannot distinguish between the various incremental sources, organic or inorganic, since virtually all of that applied is eventually converted to ammonium or nitrate form in the soil, if not so applied, and these are the forms primarily absorbed by plants.

Efforts to depict balance sheets of nitrogen input/output with the changing times have been tenuous at best. Few absolute measurements have been made on items like rainfall fallout, leaching, symbiotic and nonsymbiotic nitrogen fixation in a very limited number of ecosystems prior to the 'environmental awakening' of the 1960's, and none at all of denitrification. Thus the summation presented in Table 2 for the U.S. by Parr (1972) can be taken only at face value, but the values probably are within 25% of the actual.

Table 2. Balance sheet of nitrogen in the United States: estimated changes from 1930 to 1980 on harvested cropland (Parr, 1972).

Item	Nitrogen, millions of tons*			
	1930	1947	1969	1980
Inputs of nitrogen from				
1. Fertilizer nitrogen	0.3	0.7	6.8	12.2
2. Nitrogen fixed by legumes	1.7	1.7	2.0	2.5
3. Nitrogen fixed nonsymbiotically	1.0	1.0	1.0	1.0
4. Barnyard manure	1.9	1.3	1.0	0.9
5. Roots of unharvested portions of crops	1.1	1.5	2.5	3.0
6. Rainfall	0.8	1.0	1.5	1.8
Total	6.8	7.2	14.8	21.4
Removal of nitrogen by				
1. Harvested crops	4.6	6.5	9.5	13.0
2. Erosion	5.0	4.0	3.0	2.0
3. Leaching of soil nitrogen	4.0	3.0	2.0	1.5
4. Leaching of fertilizer nitrogen	0	0	?	1.2
5. Denitrification	?	?	?	2.2
Total	13.6	13.5	14.5	19.9

* The data are English tons (2,000 pounds/ton).

The uncertainties notwithstanding it is quite clear that fertilizer nitrogen has taken over as the predominant input, representing around 60% of the total inputs in 1980 while removal in harvested crops will account for 2/3 of the total removal. These figures correspond with respective values of 4% and 1/3 in 1930. The primary enigma in recent times and in the forward projection is with the denitrification removal. It is also this author's opinion that those responsible for the summation have been overly optimistic about the continuing reduction in eroded nitrogen with time and on the increasing amounts of symbiotically fixed nitrogen.

Much research was conducted in the developed countries for establishing effective fertilizer nitrogen practices for maximum yields under varied soil and cropping conditions from 1950 onwards, and demonstrations of the principles realized were carried to developing countries through fertilizer development programs. Only China among more advanced countries with major food requirements has resisted the trend toward predominant inorganic nitrogen use. As late as 1976 the Chinese with their populous agrarian society were deriving some 75% of the nitrogen utilized in agricultural production from organic sources (Johnson and Beemer, 1977). But even these pragmatic organic farmers will soon be satisfying a major portion of crops' needs by the inorganic nitrogen route on completion of a dozen large capacity nitrogen plants now building that will supply in the order of 3 million tons of fixed nitrogen annually.

The expanding U.S. population during the 50 years prior to 1930 was balanced by conversion of virgin lands to crop lands for providing national food and fiber needs. Since that year of maximum cultivated acres, however, there has been a reduction of almost 10% in the nation's cropland while population has nearly doubled (Figure 4). This anomaly was made possible by the advent of the chemical fertilizer industry subsequent to World War II and most particularly by the tremendous growth in fertilizer nitrogen use. The additional production afforded has not only met the needs of the nation but has contributed more than any other national product toward the balance of trade in international commerce, representing almost one-fourth of

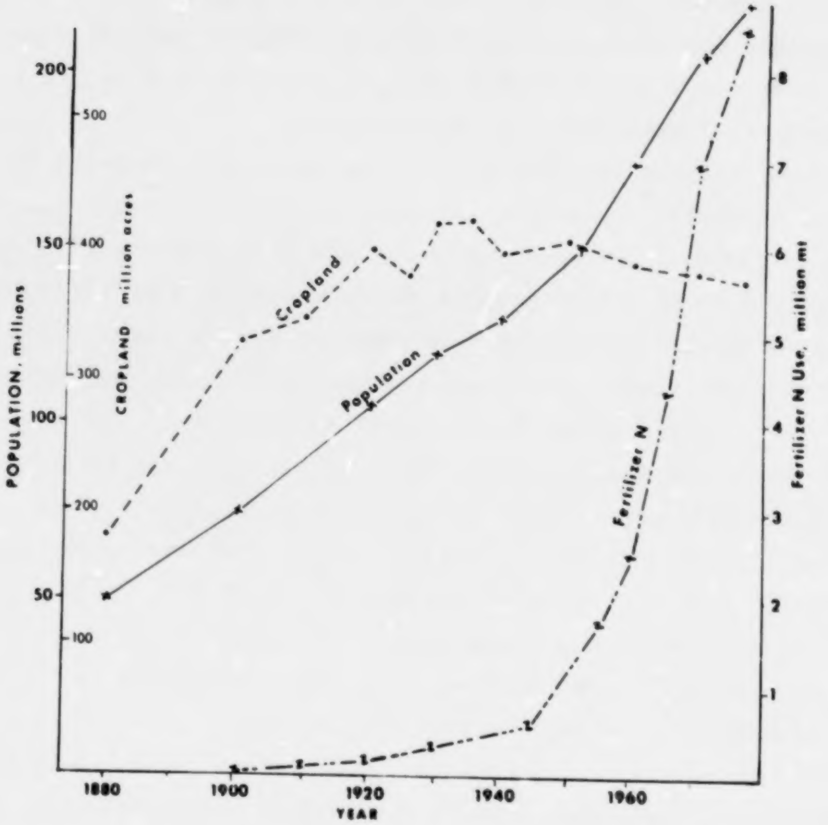

Figure 4. Relation between cultivated land, population and fertilizer nitrogen consumption in the U.S., 1880-1976. Fertilizer use is in millions of metric tons.

all U.S. exports in the late 1970's at some $22 to 24 billion dollars per annum. A very substantial portion of this income can be attributed directly to fertilizer nitrogen. Without fertilizer nitrogen the U.S. housewife would likely be paying at least twice what she does for groceries today.

Crop Response to Fertilizer Nitrogen

Response data of the type presented in Figure 5 show the dramatic effect that fertilizer nitrogen has had on world food and fiber production in the past generation. No other agricultural innovation during this period has contributed as much toward satisfying the needs of a food hungry world. Note the quadrupled average corn yields in the midwestern state of Nebraska with growth in fertilizer nitrogen consumption. Of course other technological advances contributed including improved hybrids, better control of pests, advanced methods of crop culture, expanded irrigation, etc., but the yield enhancement could not have resulted without the nitrogen input.

The success achieved by the miracle varieties of the Green Revolution was made possible by the arrival of relatively cheap fertilizer nitrogen on the scene in the 1960's according to the movement's acknowledged father (Borlaug, 1971). Without this input along with improved watering and pest control practices the new varieties were little if any better than those used traditionally by peasant farmers. Wright (1972), for example, in summarizing a series of coordinated experiments on fall-seeded spring wheats of CYMMIT origin in India noted doubled yields from the appropriate nitrogen treatment. The data also demonstrate response to notably greater rates of nitrogen by dwarf varieties than by the tall types investigated. A 120 kg/ha rate would have been most economic for the dwarf types in this study whereas no more than 60 kg/ha was needed for the tall types with respective yield levels of approximately 43 and 34 quintals/ha. He also reported similarities in response to nitrogen

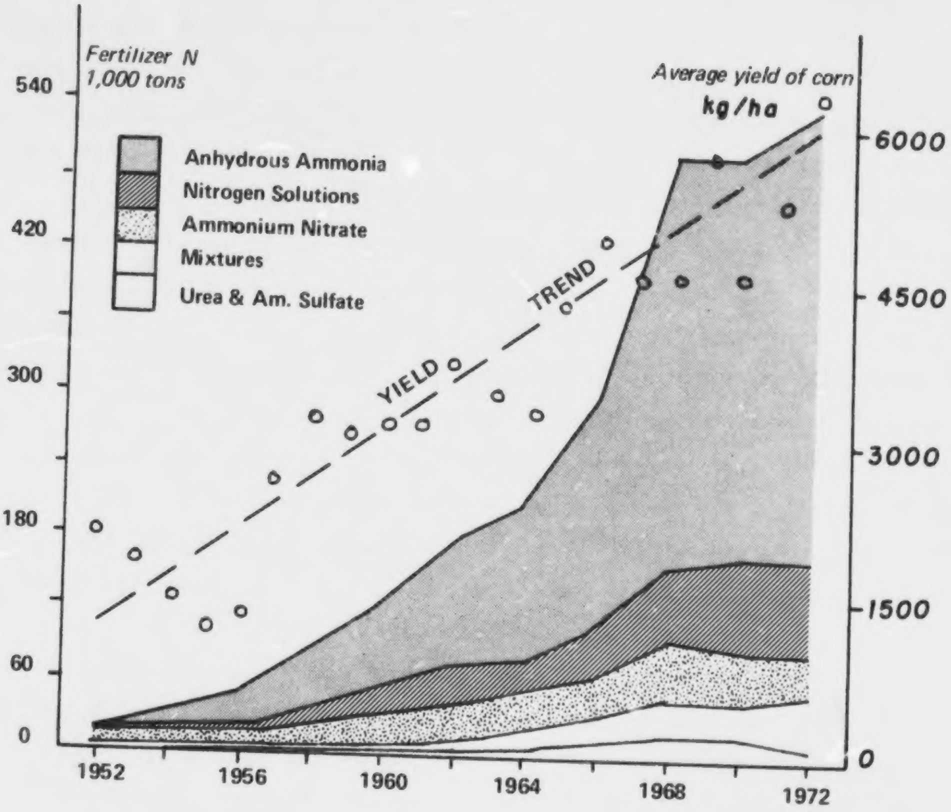

Figure 5. Average yield of corn in Nebraska in relation to statewide consumption of fertilizer nitrogen, 1952-1972.

rate in many regions where high-yielding varieties have been introduced with explanation that the wheat is generally grown on formerly desert soils, now irrigated, which were inherently of very low organic matter and nitrogen contents.

The required nitrogen for the new varieties of rice from IRRI also doubled and even trebled the practical yield potential of the traditional varieties. Note the large response of IR8 and IR20 (Figure 6) to nitrogen rates in the order of 150 kg/ha in the dry season of the Philippines and 90 kg in the wet season, in contrast with the lack of response of the Peta, a native variety selected through prior times for its performance on tropical soils of existing low fertility status (Brady, 1975).

Acceptance of the fertilizer nitrogen source was forthright in the more humid and irrigated regions with the high value/cost ratios that could be demonstrated and the minimal risk involved. Perhaps the greatest deterrant was the old-line agronomist who insisted that phosphating the legumes to harvest a large nitrogen yield was still the logical route to follow. Farmer acceptance was much slower, however, in the semi-arid cropping regions where taller, greener plants didn't necessarily portend greater yields of grain. With moisture supply for the crop ever critical it was a matter of balancing the highly growth stimulating nitrogen practice against soil moisture storage and rainfall probabilities. Thus, data of Ramig and Rhoades (1963) suggested likely economic response of winter wheat to 22 kg nitrogen/ha for each 7 cm stored moisture in the rooting profile at planting of soil with nominal nitrogen capacity in a region of 15 cm growing season rainfall. Figure 7 also demonstrates the critical need for effective nitrogen management practice in dry-farming wheat production of western Nebraska where use of only slightly more than optimum nitrogen rates can have depressive effects on grain yield and economic return. Higher than economic rates for grain sorghum did not depress yields, however, and for the reason that summer side-dress application

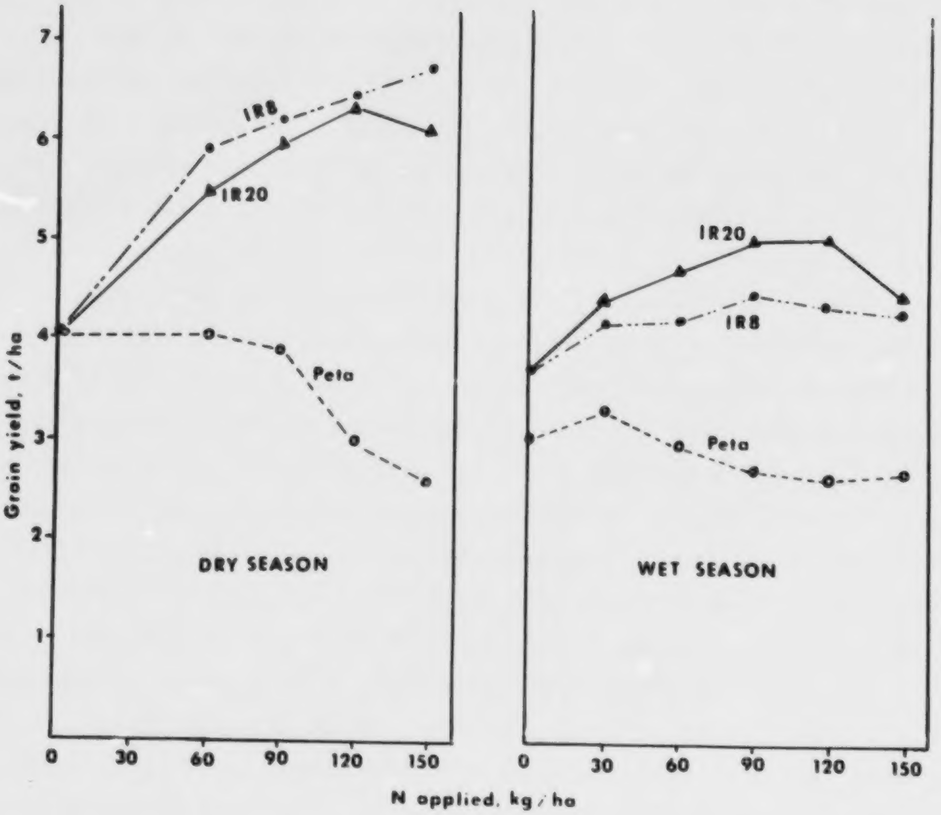

Figure 6. Effect of nitrogen level on grain yield of three rice varieties during wet and dry seasons at four Philippine locations averaged over six cropping seasons, 1968-73 (Brady, 1975).

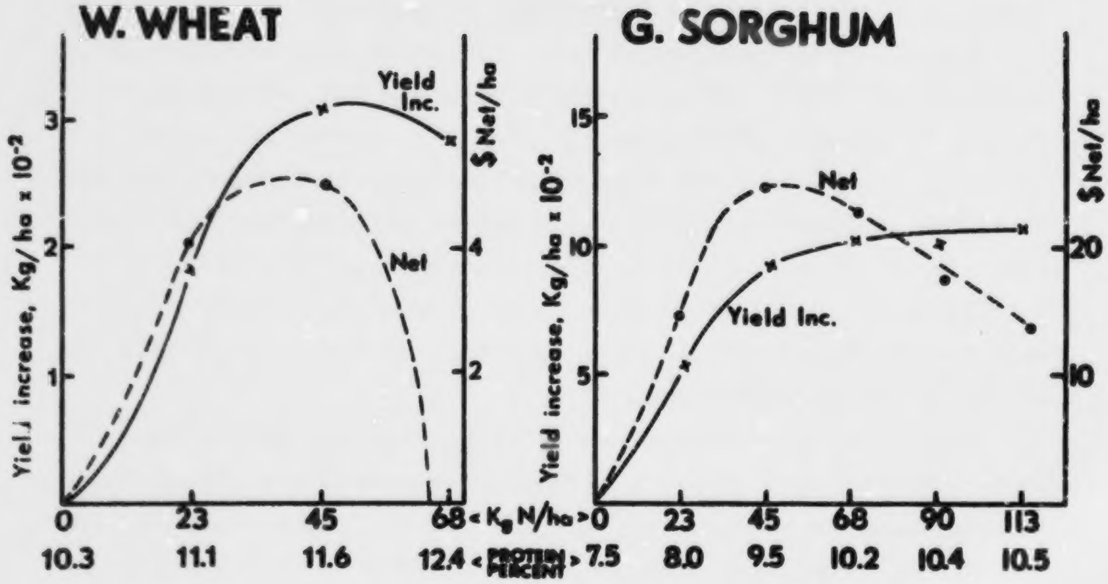

Figure 7. The response of winter wheat and grain sorghum to fertilizer nitrogen in dryfarming, average of 54 experiments with spring applied nitrogen on winter wheat and 10 experiments with side-dressed nitrogen on grain sorghum, Nebraska, 1959-1967.

was employed which did not stimulate excessive vegetative growth.

It has been estimated that fertilizer nitrogen is responsible for 33% of the total agricultural production in the U.S. today (CAST, 1977). In some regions where crops of high nitrogen demand are grown and where native levels of soil phosphorus and potassium are high, as in the western corn belt of the U.S., fertilizer nitrogen accounts for an even greater portion, perhaps as much as 50% of current production.

Improving the Efficiency of Fertilizer Nitrogen

The 2% of the total natural gas consumption that is presently involved in fertilizer nitrogen production in the U.S. affords manifold returns in calories of food energy. Few will contest its worth. There may be contention by the turn of the century, however, when the total energy demand for fertilizer nitrogen per annum may reach 10% of U.S. gas consumption, with total fertilizer nitrogen production costs of $30 billion.[2/] Should this indeed come to pass the country's citizenry will demand maximum efficiency in use of the product irrespective of environmental considerations.

Rate. The tremendous impact of fertilizer nitrogen on world crop production notwithstanding, there is a generally recognized need for improving the efficiency of its use. In dry regions the rate employed must be balanced with the moisture likely to be available throughout the growing season so that overstimulation of early growth does not result in yield reduction at season's end. Even with plentiful moisture it is necessary to keep nitrogen supplied to the plant within reasonable bounds for small grains if lodging and consequent yield loss are not to occur, for sugar crops so that excessive vege-

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[2/] Frederick Ausubel. Increasing crop yield by increasing the amount and efficiency of biological nitrogen fixation by means of recombinant DNA technology. Dept. of Biology, Harvard Univ.

tative growth does not take the place of sugar set, for tree fruit crops to assure good appearance and keeping quality of the fruit, and for many crops in the control of stalkrots and other plant diseases.

Determination of an effective rate requires a series of fertilizer nitrogen rate experiments with any crop under consideration. Commensurate measurements must be made on soil nitrogen availability from native and residual fertilizer nitrogen sources in the crop rooting zone so that the acquired response data can be used for predictive purposes. A subsequent paper treats the soil nitrogen evaluation matter in detail.

Excessive rates of fertilizer nitrogen have been employed locally, especially for some of the high-value vegetable and fruit crops and particularly in association with irrigated culture. But heavier than necessary quantities have been employed as well in the Corn Belt of the U.S. Illinois, for example, applied 29% more nitrogen than was harvested in crops in 1970 (Welch, 1972), and Nebraska used nearly 45% more nitrogen than was harvested in 1968 (Figure 8). Such application rates totally disregard nitrogen release that can be expected from soil reserves, fallout in rain and the various natural fixation processes. The excesses can be attributed in large part to the overproduction and depressed market value of fertilizer nitrogen in the late 1960's, and do show how farmers will insure against any possibility of shortage if the cost for nitrogen is low.

Time and Method of Application. Appropriate time and method of applying fertilizer nitrogen is determined especially by growth characteristics of the crop, climatic factors, soil properties and nature of the nitrogen carrier. As a very general rule, efficiency in use is enhanced if the major portion of the applied nitrogen can be programmed to become positionally available to the root system just prior to the period of maximum need by the crop. At the same time the active root system limits the amount that can escape the root

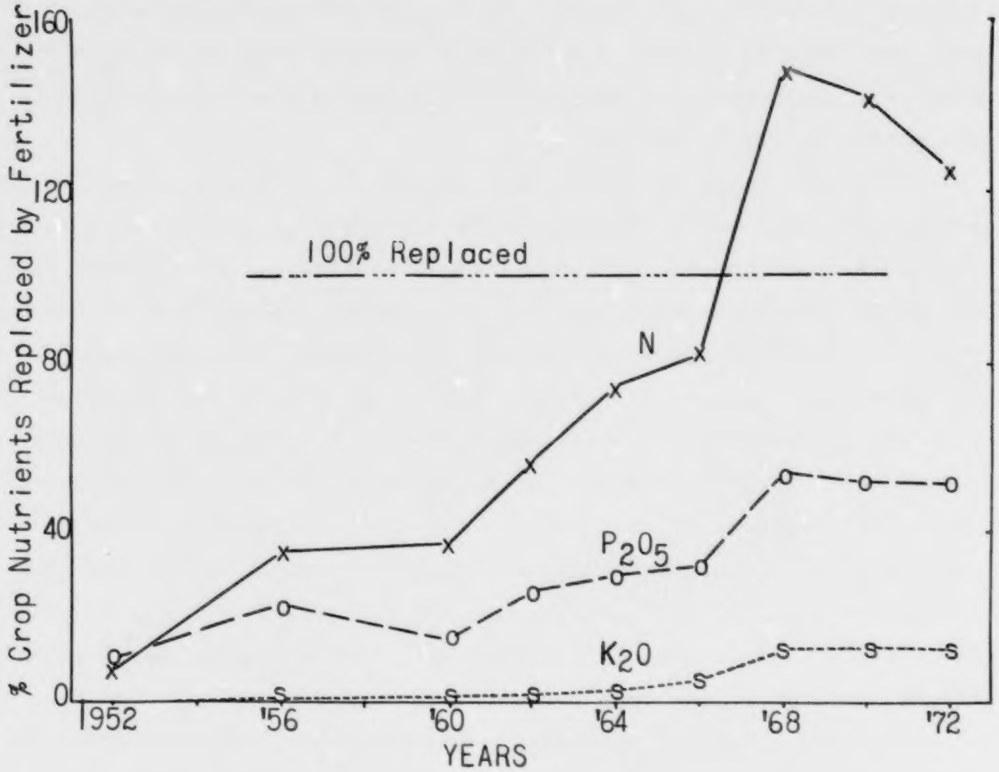

Figure 8. Proportion of harvested crop nutrients replaced by fertilizer, Nebraska, 1952-72. In Olson, R. A. 1974. Fertilizer nitrogen: Mainstay for agriculture. Farm, Ranch and Home Quart., vol. 20:20-22. Univ. of Nebr. Coll. Agr., Lincoln, Nebraska.

zone by leaching. So it is that summer sidedressing of corn under irrigated and humid cropping conditions commonly results in most efficient response to the nitrogen by the current crop and greatest residual remaining in the soil for use by the next crop (Olson et al, 1964a; Welch et al, 1971; Herron et al, 1971; Miller et al, 1975). Good evidence exists, furthermore, that nitrogen taken up relatively late in the growth period is channeled more directly to grain with less immobilization in the vegetative portions of the crop (Gass et al, 1971). A crop like cool season grass grown for seed, on the other hand, must absorb a major portion of the nitrogen required very early in the growing season to obtain maximum seed yield. Timing of nitrogen availability is especially critical for a crop like sugar beets which requires a high level of nitrogen availability throughout the first 3/4 of the growing season but essentially a deficiency of nitrogen during the final quarter to assure maximum sugar set (Alexander et al, 1954; Rhoades and Harris, 1954).

In the very humid regions and with very sandy soils of low water holding capacity it may prove necessary to apply nitrogen in two or more increments to reduce the leaching potential to acceptable levels. For example, it has become common practice to apply nitrogen in increments throughout the irrigation season where corn is now grown on sands under center pivot irrigation systems.

The chemical carrier of the nitrogen source will often dictate what is effective time and method of application. Anhydrous ammonia, for example, is equally effective when fall applied after soil temperature has dropped below 40F to that summer sidedressed for a summer crop like corn, while a carrier like ammonium nitrate could suffer serious leaching losses during the late fall, winter and early spring months from fall application. Placement method is correspondingly dictated by the chemical carrier involved, a prime example being the necessity of incorporating urea to limit ammonia volatilization losses.

Environmental Considerations. As with other chemicals introduced into agricultural production systems, virtually all of the attendant research was directed toward the impact on the target crop without regard for the surrounding ecosystem or those adjacent. It required a 'Silent Spring' (Rachel Carson, 1962) to focus international attention on effects that the chemical treatment might have beyond the intended target. The ensuing years have brought forth a modified perspective of most economic return from the fertilizer nitrogen with minimum contamination of the environment (Viets and Hageman, 1971). Many facts have been elaborated concerning nitrogen in the environment during the interim while at the same time much fantasy has been written by self-styled environmentalists clouding the issue. Debate will continue on the magnitude of environmental contamination by fertilizer nitrogen until much corroborative research has been completed across many and varied ecosystems. In view of the number of documented situations where nitrogen from fertilizer sources has contaminated ground water (Ward, 1970; Bingham et al, 1971; Ayres and Branson, 1973; Muir et al, 1973), however, it becomes mandatory that farmers derive maximum efficiency in crop use to minimize the pollution hazard and, at the same time, serve their own best economic interests.

Converse to the detrimental features to be treated in subsequent papers, fertilizer nitrogen use provides a number of environmental benefits. Most obvious, of course, is the enhanced agricultural productivity afforded as previously elaborated. After all, what environmental quality is more important to man than a sufficient food supply? Related is the enhanced protein and vitamin contents of food grain crops that commonly accompany nitrogen fertilization for optimum yield. A second environmental benefit is the improvement in the physical, biological, and chemical character of soils from judicious fertilizer use, coming from the larger organic return and greater root proliferation of thrifty, well-fertilized crops. Farming efficiency is improved by fertilizer nitrogen, thirdly, allowing increased production on fewer acres while permitting retirement of erosive, low productivity land to permanent vegetation (again, see Figure 4). Fourth, by producing greater vegetative and root growth, applied nitrogen is one of the most effective of all erosion control measures. Finally, of special significance in regions of limited moisture availability is the enhanced water use efficiency with provision of sufficient nitrogen for the crop. Conservancy in water results when yield is increased by added nitrogen since the added yield is accomplished with only a little more water consumed than is used by the non-treated crop (Viets, 1962; Olson et al, 1964b).

Influence of Applied Nitrogen on Crop Quality

The most commonly recognized benefit of applied nitrogen to crop quality is that of increased protein content of the food grains, of especial import to the developing countries where around 2/3 of the protein in the daily diet comes from cereals in contrast with the 1/5 from that source in the U.S. (F.A.O., 1968). Much of the past research on production practices for these crops has concerned itself with grain yield, and little more than casual notice was paid to the crude protein in the grain. The milling and baking industries of the developed countries orient their purchasing practices to specific regions where soil and climatic environment produce the protein levels required. Recently, however, world protein needs have become increasingly apparent. Moreover, within individual countries some regions of traditionally high protein grain can no longer supply the expected product by reason of soil depletion and protein dilution imposed by higher yielding varieties.

There was no general agreement on benefit to protein of cereal grains from nitrogen fertilization until the period following World War II. The majority of fertilizer trials prior to that time employed very low rates of nitrogen, as in the order of 10 to 30 kg/ha, for which today we would predict limited if any protein increase. Thus, field experiments at the Rothamsted experiment station through the late 1930's showed no clear-cut effects from nitrogen fertilizaion on grain protein (Russel and Watson, 1940). Most recent literature on fertilization of cereals, however, shows increasing protein content of the grain with increasing nitrogen rate, at least above some minimal rate which increases yield at the expense of protein. The rate of protein increase from that point onward is usually a near straight line function with rate of nitrogen until some protein maximum for the varieties employed is reached, normally well in excess of the rate required for maximum grain yield (Figure 9).

Most studies indicate that applying nitrogen in the spring for winter wheat results in a higher grain protein content than results from fall application with a slight yield advantage as well in the semi-arid wheat producing areas (Hucklesby et al, 1971; Olson et al, 1964a). Protein percentage tends to become progressively higher as time of nitrogen application is delayed, providing precipitation is received to carry the nitrogen to the root system. With foliar treatments nitrogen application may be delayed so that the entire expressed nitrogen effect is on protein and not at all on yield (Finney et al, 1957). Protein increases ranging from 2 to 4% have thus been noted from spraying urea solution in three increments to a modest total rate of 50 kg nitrogen/ha on winter wheat, the most benefit deriving from the treatment made at early bloom state (Sadaphal and Das, 1966).

Delayed application of economic rates of nitrogen for corn has also been more effective for grain and protein yields than earlier treatment (Figure 10). Submerged rice as well benefits more from fertilizer nitrogen applied during the growing season than at planting or before in terms of total nitrogen uptake and utilization for grain protein (Beachell et al, 1972; DeDatta et al, 1972; FAO/IAEA, 1970). Application shortly before primordial initiation has proved optimum for maximum protein production with a given nominal nitrogen

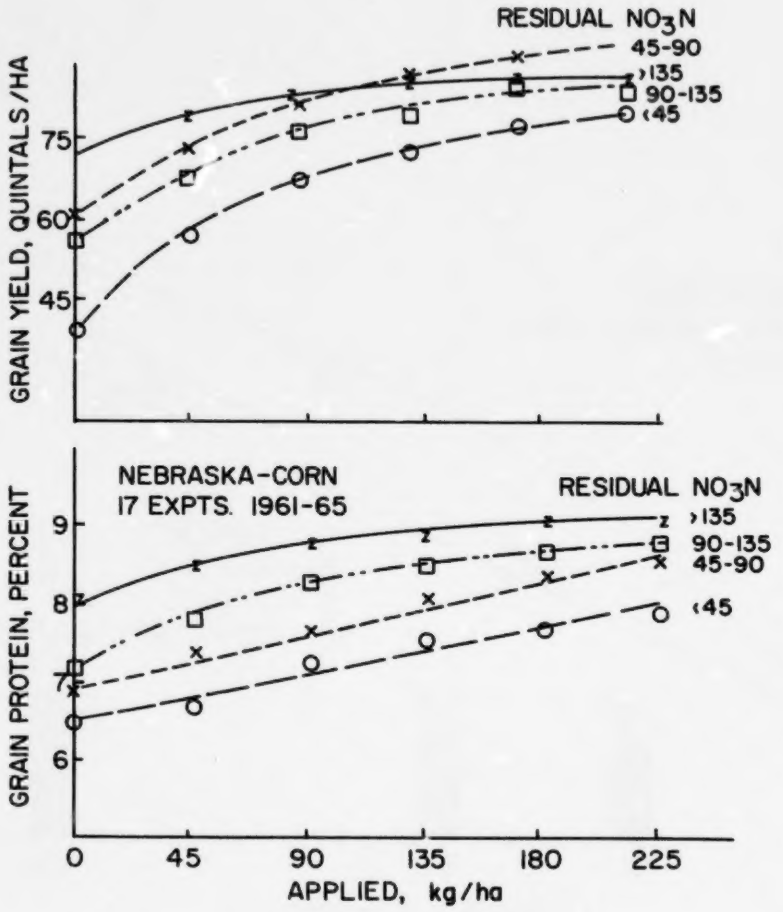

Figure 9. Grain yield and protein content of irrigated corn as influenced by soil residual nitrate nitrogen and applied fertilizer nitrogen in 17 Nebraska field experiments, 1961-65.

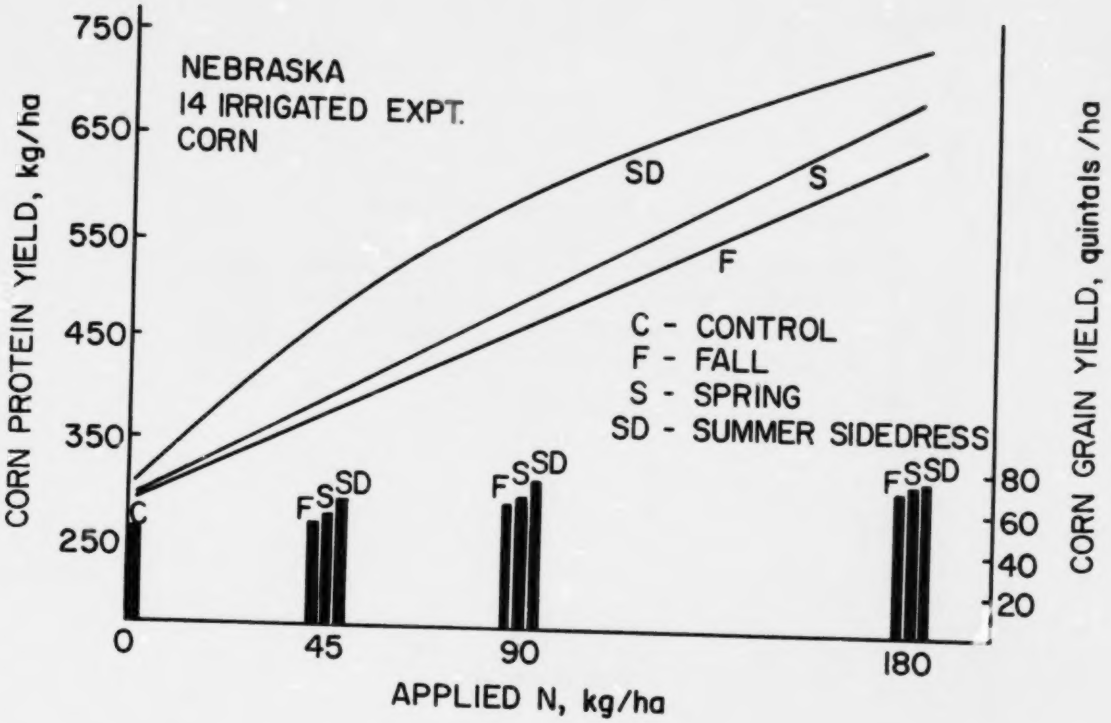

Figure 10. Average crude protein and grain yields in 14 irrigated corn experiments in Nebraska as influenced by time and rate of fertilizer nitrogen application (F = fall, S = spring, SD = summer sidedress application of nitrogen fertilizer).

rate for representative soils having sufficient nitrogen reserve to get the plant well started. Thus it appears that for all cereals the latest possible application of nitrogen commensurate with crop stage of development and growing conditions that permit ready nitrogen uptake, provides for maximum protein yields.

The leaf crops such as lettuce and spinach are markedly enhanced in succulence and marketability when produced with sufficient nitrogen. The grower in this case must employ the quantity of fertilizer nitrogen required for this attribute commensurate with keeping the nitrate in the plant tissue within acceptable levels for human nutrition. On the other side of the ledger there is documentary evidence of decreased shelf life of various vegetables and fruits, of increased contents of undesired free amino acids, of decreased ascorbic acid and potassium contents of produce, and of loss in flavor of some commodities with excessive use of fertilizer nitrogen (Schuphan, 1972).

Other quality aspects, pro and con, related to fertilizer nitrogen exist for most commercial crops exceeding the bounds of this paper for elaboration. Suffice it to say, the nitrogen nutritional needs of the world's present man and animal population would be woefully undersupplied without an inorganic fertilizer input.

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NITROGEN FORMS AND CYCLING IN RELATION TO WATER QUALITY[1/]

Ronald G. Menzel[2/]

ABSTRACT

The intended use of water determines the significance of various forms of nitrogen in water quality. Upper limit concentrations have been recommended for several forms of nitrogen in water used for public water supplies, freshwater aquatic life, marine aquatic life, and agricultural uses. The concentrations are lowest for cyanide, and increase in the order ammonium, nitrite, and nitrate. Nevertheless, nitrate is more often of concern in water quality than are the first three forms of nitrogen.

The recommended upper limit concentration for nitrate-nitrogen in public water supplies is 10 mg/l. The actual concentration in ground water exceeds this limit in some areas, and may be increasing. Because of the slow movement of ground water, increasing concentrations may take many years to correct.

Nitrate is depleted from surface waters by plant uptake, and may cause excessive growth of algae or other plants. This is seldom a problem in streams, but may become one in impoundments, estuaries, and near-shore ocean areas.

Much of the total nitrogen in surface waters occurs in particulate or dissolved organic forms. Plant growth is often excessive when the total nitrogen concentration exceeds 0.5 mg/l if other nutrients are adequately supplied and growth conditions are favor-

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able. Nitrogen supply limits growth in many ocean areas and in a few lakes, mainly eutrophic ones. Nitrogen fixation by blue-green algae tends to correct nitrogen deficiencies. Denitrification by bacteria tends to correct excesses. These features indicate that high nitrogen concentrations may be an effect, rather than a cause, of eutrophication.

INTRODUCTION

Nitrogen occurs in a variety of chemical forms in all natural waters. Some are readily soluble plant nutrients, such as nitrate and ammonium ions. Some may be highly toxic, such as cyanide or nitrite, which form occasionally during animal or plant decomposition. Some are organic compounds, varying in complexity from simple amino acids to nucleic acids. A few pesticide compounds contain nitrogen.

The kinds and amounts of nitrogen forms in water are closely connected with the growth and decay of organisms. Transformations among nitrogen compounds are often carried out biologically. Since biological activity depends on such factors as light intensity, flow velocity, and depth of water, it will be convenient to discuss water quality separately for ground water, streams, impoundments, and marine environments. In each environment, the prevalent forms, transformations, and important water quality effects of nitrogen are somewhat different.

RECOMMENDED UPPER LIMIT CONCENTRATIONS

Dissolved forms of nitrogen have the most direct effects upon water quality. For some of these forms, upper limit concentrations in water have been recommended, always in relation to specific uses of water. The limits recommended in Water Quality Criteria 1972 by a Committee of the National Academy of Sciences - National Academy of Engineering are summarized in Table 1.

Table 1. Summary of water quality criteria for common forms of nitrogen. (from Water Quality Criteria 1972).

Form of nitrogen	Type of Water Use*			
	I	II	III	IV
	Recommended Maximum Concentration (mg N/liter)			
Nitrate	10			100$^+$
Nitrite	1			10$^+$
Ammonium	0.5	0.02‡	0.01‡	
Cyanide	0.2	0.005‡	0.005‡	

*
I Public water supplies.
II Freshwater aquatic life.
III Marine aquatic life.
IV Agricultural uses, including livestock watering and irrigation.

$^+$ For livestock water.

‡ Nitrogen as unionized ammonia or hydrocyanic acid.

The highest concentrations in Table 1 are shown for nitrate in public water supplies and for livestock watering. The hazard from nitrate arises principally from its possible reduction to nitrite in the digestive tract, especially in infant humans and ruminant animals. Upper limit concentrations were not set for water supporting aquatic life. However, the Committee concluded that as little as 0.36 mg/liter of nitrogen as nitrate could produce enough organic matter to exhaust the oxygen content of marine water. They considered this an excessive concentration.

The limiting concentrations of nitrite are based on its effect in the blood stream when water is drunk by humans or animals. Nitrite combines with the hemoglobin of red blood cells so that they cannot transport oxygen. Nitrite concentrations are too low in most natural waters to be of concern for freshwater or marine aquatic life. Therefore, upper limit concentrations of nitrite were not set for these water uses.

Ammonia is quickly formed from all kinds of animal excrement, and dissolves in water to form ammonium ions. The recommended upper limit in public water supplies is based on the facts that ammonium may indicate other types of pollution, and that it requires additional chlorination for water treatment. Fish are sensitive to unionized ammonia, which means that the toxicity of a given ammonium concentration increases rapidly as pH values increase above 8. The percentage of unionized ammonia is 3% at pH 8, 10% at pH 8.5, and 30% at pH 9.

Cyanide is highly toxic to fish, the toxicity increasing rapidly at pH values below 8. This indicates that the hydrogen cyanide molecule is the effective toxicant. The Committee indicated that industrial wastes were the principal sources of cyanide in waters. Some forage grasses and other plants contain hydrogen cyanide after freezing or other stresses (Heath et al, 1973). Such plants may at times release hydrogen cyanide into nearby bodies of water.

The Committee considered many other compounds in relation to water quality. Among the nitrogen containing compounds considered were a number of pesticides. Most nitrogen containing pesticides degrade in water, eventually forming ammonium and then nitrate. However, the amounts of these compounds added to water in pesticides are very small compared with other sources or forms of nitrogen.

NITROGEN IN GROUND WATER QUALITY

Nitrate is the major form of nitrogen in ground water. Three extensive ground water surveys are summarized in Table 2.

Table 2. Amounts of nitrogen found as nitrate, ammonium, and nitrite in ground water samples.

Source of water and reference	Number of samples		Nitrogen (mg/l) in the form of		
			Nitrate	Ammonium	Nitrite
Swedish wells, Wiklander (1977)	179	Ave.	3.2	0.1	0.016
		Max.	177.0	56.5*	13.4*
Wisconsin dairy wells Crabtree (1972)	59	Ave.	9.9	2.0	0.02
		Max.	27.3	5.1	0.30
Nebraska irrigation wells Olson et al (1973)	364	Ave.	3.1	---	No analysis
		Max.	---	0.34	

* Not included in the average.

They show average concentrations ranging from 3 to 10 mg/l of nitrogen as nitrate. Generally less than 1 mg/l of nitrogen occurs as ammonium and less than 0.1 mg/l as nitrite. In Sweden exceptionally high concentrations of all forms of nitrogen were found in two old unused wells close to farm houses. Occasional high concentrations of nitrite in Wisconsin occurred mostly in shallow dug wells immediately after heavy rains.

Nitrate appears to be quite stable in ground water. The filtering action of soil removes bacteria and most organic matter so that there can be little biological activity that might remove or transform the nitrate. Thus, in the Arroyo Grande Basin near San Luis Obispo, Stout and Burau (1967) followed seasonal increments of nitrate down several cropped soil profiles.

Nitrite and ammonium, on the other hand, are not as stable as nitrate. Feth (1966) discussed two California studies showing that neither ion travelled far after injection into shallow ground water. The presence of nitrite or ammonium in ground water usually indicates surface contamination.

Concentrations of nitrate higher than the recommended limit for public water supplies have been observed in ground water in California and several midwestern states. Ward (1970) discussed two California surveys, the first a state-wide survey conducted in 1954 and 1955. Of 541 wells sampled, 33 had water containing more than 10 mg/l of nitrate-nitrogen. These high nitrate wells were located mostly in the south central and interior basins of Southern California. During 1961, a second survey was made of 100 domestic wells in each of eight Southern California counties. Samples were collected in January, April, July, and October. Of the 800 wells, the number containing water with more than 10 mg/l of nitrate-nitrogen ranged from 49 in January to 34 in July. Minor forms of nitrogen were also determined and were generally found in the following decreasing order: ammonium, organic nitrogen, and nitrite.

In addition to the concern about its effects in drinking water, excessive nitrate in ground water moves into surface waters in some areas. Biggar and Corey (1969) estimated that 45% of the total nitrogen entering Lake Mendota in Wisconsin moved as nitrate in groundwater. Movement of nitrate from irrigated areas into the San Luis drain in the San Joaquin Valley has received much study (Miller and Smith, 1976). The role of nitrate in eutrophication of streams, lakes, and estuaries will be discussed in subsequent sections.

Nitrate concentrations in ground water are increasing in some areas. Adriano et al (1971) noted concentrations averaging 54 mg/l of nitrate-nitrogen at the top of the water table in the dairy area between Riverside and Los Angeles. Deeper domestic wells in the same area yielded water averaging only 6 mg/l of nitrate-nitrogen. The concentration should increase as the nitrate at the top of the water table mixes into the body of ground water. In a study of 379 Nebraska irrigation wells, Olson et al (1973) sampled the same wells in 1961 and 1971. The average concentration of nitrate-nitrogen increased from 2.4 to 3.1 mg/l in 10 years. In a study of one fertilized agricultural site in southern England Young et al

(1977) projected an increase in nitrate-nitrogen concentration in the ground water from 4 mg/l in 1970 to 14 mg/l in 2000.

Viets and Hageman (1971) pointed out that the hazard from nitrate contamination depends greatly on the residence time of ground water. Where residence time is short, as in some humid regions, nitrate would probably not accumulate and corrective measures could be effected rather quickly if it did. Where residence times are long, as in some Midwestern and Plains States, nitrate contamination might not become evident for decades after the process had started. An equally long time would be required to correct it.

NITROGEN IN STREAM WATER QUALITY

Flow is an essential characteristic of streams that is highly relevant to water quality. Nitrogen and other nutrients do not accumulate in flowing streams, but are only detained temporarily in sediments and vegetation. Typical streams change from fast, turbulent upstream flows to slower, sometimes laminar downstream flows (Hynes, 1970). Typical vegetation changes accordingly from attached algae and other plants adapted to running water to drifting algae and plants that root in fine sediments. Dissolved oxygen concentrations are always near saturation upstream, but may sometimes decrease markedly downstream.

The forms of nitrogen in streams reflect those in the source waters, as modified by biological activity. In the Kaskaskia River of Illinois, Smith et al (1975) reported concurrent nitrate-nitrogen concentrations in the river and a major tile outlet into it over a 30-month period. Monthly average concentrations ranged from 2 to 12 mg/l in the river and from 5 to 12 mg/l in the tile outlet. The concentrations in the river and tile waters were nearly equal except during July, August, and September, when concentrations in the river were much lower than in the tile outlet. Relatively low flows, averaging about 30 l/sec, and warm water temperatures favored biological activity in the river during these months. Feth (1966) reported that in most United States streams, nitrate concentrations were highest between December and April, and lowest between May and November. Other forms of nitrogen were not reported in these studies.

In many upstream waters, the forms of nitrogen correspond well with those in land surface runoff. This might be expected considering that streams generally carry suspended soil particles and decomposing plant residues. Similar microbial transformations of nitrogen occur in moist soils and in streams. Typical concentrations of different forms of nitrogen in small streams and in land surface runoff are summarized in Table 3. Although the examples are few and not directly comparable, the dissolved nitrogen in streams is predominantly nitrate whereas runoff sometimes contains more nitrogen as ammonium than as nitrate. Another distinguishing characteristic of land runoff is the occasionally high concentration of nitrogen carried on soil particles. Concentrations of particulate nitrogen in streams decrease as the larger soil particles settle out of the flowing water.

Removal of nutrients by vegetation growing in and along streams has not been studied extensively. McColl (1974) showed that ammonium, but not nitrate, was quickly removed by attached filamentous algae and entrapped sediment in a small New Zealand forest stream. Manny and Wetzel (1973), working on an 8 km reach of Augusta Creek in Michigan, showed daytime decreases in nitrate, ammonium, and labile forms of dissolved organic nitrogen compared with their nighttime concentrations. The relative decreases were about 20% for nitrate, 50% for ammonium, and 30% for organic nitrogen. Concentrations of all forms generally decreased slightly in the downstream direction. Manny and Wetzel concluded that ground water influx and vegetative uptake of the nitrogen compounds produced the observed changes.

Microbiological transformations of nitrogen occurring in streams

Table 3. Amounts of nitrogen found as nitrate, ammonium, nitrite, dissolved organic, and particulate forms in land runoff and streams.

Source of water and reference	Typical ranges of nitrogen concentration (mg/l) in the form of				
	Nitrate	Ammonium	Nitrite	Dissolved Organic	Particulate
Upper Danube Vollenweider (1971)	1.12-1.43	0.25-0.51	0.025-0.04	--*	--
Nebraska streams Olson, Seim, and Muir (1973)	0.1-1.1	0.04-0.36	--	--	--
Augusta Creek, Michigan Manny and Wetzel (1973)	0.63-0.98	0.02-0.10	--	0.22-0.34	--
Runoff, aspen-birch forest Timmons et al (1977)	0.08-0.33	0.07-0.55	--	--	0.97-2.33**
Runoff, southern pine forest Schreiber, Duffy, and McClurkin (1976)	0.03-0.10	0.72-1.52	--	--	--
Runoff, Oklahoma cropland Olness et al (1975)	0.3-1.4	0.05-0.20	0.01-0.04	--	1.4-3.7**
Runoff, Iowa cropland Schuman et al (1973)	0.9-4.5	0.8-3.2	--	--	0.4-100
Runoff, Minnesota cropland Burwell, Timmons, and Holt (1975)	0.2-5	0.3-1	--	0.3-1.2	0-180

* Dash means this form of nitrogen was not determined.

**These values include dissolved organic nitrogen.

have been nicely summarized by Rhenheimer (1974). Many of the same bacteria that are present in soils convert organic nitrogen to ammonium, ammonium to nitrite, or nitrite to nitrate. Numerous bacteria can convert organic nitrogen to ammonium. They multiply rapidly whenever proteins become available, as in additions of sewage or manure. The optimum temperature for this conversion is 30-35C, but it continues even under ice with increased bacterial populations compensating somewhat for their reduced activity. Specialized bacteria convert ammonium to nitrite, and nitrite to nitrate. The dissolved oxygen required for these conversions is usually available in streams. Activity of the nitrite-forming bacteria practically ceases at water temperature below 10C. Activity of the nitrate-forming bacteria is not as sensitive to low temperatures but may be more sensitive to low oxygen supply. Microbial denitrification (reduction of nitrate to gaseous nitrogen) and nitrogen fixation (reduction of gaseous nitrogen to ammonium) do not appear to be important in stream waters.

The transformations of nitrogen in stream sediments and in streambank soils in relation to water quality have not been studied extensively. For small streams, these transformations may be more significant than those that take place in the water. Edwards and Rolley (1965) showed high rates of denitrification of sediment cores from English rivers.

Undesirable plant growth in streams is not easily related to excessive concentrations of nitrogen compounds. In various cases, growth may be limited by shading, turbidity, flow conditions, or low concentration of other plant nutrients. Patrick (1973) found that concentrations of nitrogen, mainly as nitrate, up to 2.5 mg/l did not produce excessive algae growth. Similarly, Moore (1977) found that density of attached filamentous algae was only slightly influenced by nitrate-nitrogen concentrations up to 5 mg/l in a small English stream. The density of algae attached to mud or stones was controlled mainly by light intensity and water velocity. None of the factors, temperature, pH, nor levels of dissolved phosphate, silica or nitrate had any major effect on the density of algae attached to plants.

NITROGEN IN IMPOUNDMENT WATER QUALITY

Impoundment of water in lakes, ponds, or reservoirs permits greater recycling of nitrogen and other nutrients than is possible in streams. Through the growth and decay of organisms, the supply of nitrogen compounds in an impoundment may be reused several times during a year. Algae and other aquatic plants absorb most of the nitrate and ammonium from impoundments when growing conditions are favorable. Dead plant and animal bodies settle to the bottom, where microbiological decay gradually releases ammonium back into the water. In deep water the cycle may be completed within the water column.

Thus, the forms of nitrogen in impoundments reflect biological activity to a greater extent than do the forms of nitrogen in streams. The forms vary with season and oxygen depletion of the water. This subject is discussed in an informative chapter on nitrogen in "Limnology" by Wetzel (1975).

Organic forms of nitrogen predominate in many impoundments. Concentrations are usually in the range of 0.1 to 1 mg/l of organic nitrogen. In summer, much organic nitrogen is contained in algae and other organisms. However, on the average, more organic nitrogen occurs as dissolved amino acids, peptides, or more complex compounds excreted from living or decaying organisms. The amount of dissolved organic nitrogen in impounded water is often greater than the amount of inorganic nitrogen.

The forms of inorganic nitrogen in impoundments depend greatly on the presence of dissolved oxygen. When oxygen is exhausted, as it may be in deep unmixed water or under prolonged ice cover, nitrate is reduced to nitrite and gaseous nitrogen. Ammonium released from decaying organic matter under these conditions is not oxidized. More than 5 mg/l of ammonium-nitrogen may accumulate near the sediments in eutrophic impoundments. Concentrations of nitrite-nitrogen rarely exceed 0.1 mg/l. These accumulations of ammonium and nitrite are alleviated in spring and fall in most impoundments when the surface and deep waters reach the same temperature and mix freely. At such times the deep waters receive oxygen, which allows ammonium and nitrite to be oxidized to nitrate. Nitrate is usually the predominant form in surface water with concentrations in winter and early spring ranging up to 4 mg/l of nitrate nitrogen. However, summer algae growth reduces the concentrations in many impoundments to less than 0.01 mg/l of nitrate-nitrogen.

The water leaving an impoundment usually contains less nitrogen than the water entering it. Olson et al (1973) found nitrate-nitrogen concentrations of 0.5-1.2 mg/l entering, and 0.0-0.1 mg/l leaving three Nebraska reservoirs. Ammonium-nitrogen concentrations were about the same entering and leaving, 0.14-0.31 mg/l. Similar results were reported from a central Missouri reservoir by Rausch and Schreiber (1977). They also measured the retention, or trapping of phosphate and sediment in the reservoir. Thus, impoundments may be useful in management of several aspects of water quality.

The major input of nitrogen into reservoirs is usually in streamflow. But, in some lakes, such as Lake Mendota, groundwater inflow is the main source of nitrogen (Biggar and Corey, 1969).

Nitrate or ammonium dissolved in rainfall and nitrogen fixation are other natural sources of nitrogen that may be important in some impoundments. The total nitrogen dissolved as nitrate and ammonium in rainfall usually ranges from 0.5 to 2 mg/l (Junge, 1958). These concentrations are similar to those found in streams, and higher than those found in most impoundments. Assuming a concentration of 1 mg/l and rainfall amounting to 80 cm annually, the deposition of dissolved nitrogen amounts to 8 kg/ha. Biggar and Corey (1969) estimated that rainfall contributed 17% of the nitrogen entering Lake Mendota.

Nitrogen fixation and denitrification seem to play an enhanced role in the nitrogen balance of eutrophic impoundments. Annual fixation of nitrogen by blue-green algae amounted to 0.1 g/m^2 in moderately productive Lake Windermere of England (Horne and Fogg, 1970) and 0.4 g/m^2 in eutrophic Clear Lake of California (Horne and Goldman, 1972). These amounts were less than 1% of the total nitrogen input to Lake Windermere, but were 43% of the input to Clear Lake. Nitrogen fixation provided 14% of the nitrogen input to moderately eutrophic Lake Mendota in Wisconsin (Biggar and Corey, 1969). Denitrification is carried out by many bacteria in the absence of dissolved oxygen. Brezonik and Lee (1968) estimated that denitrification removed 11% of the nitrogen input to Lake Mendota. Significant losses of nitrogen, amounting to 44% of the nitrogen regenerated from decaying organic matter, were noted in Lake Erie and presumed to be a result of denitrification (Burns and Ross, 1972). Hutchinson (1957) believed that significant ammonia volatilization from some lakes was possible during the fall mixing of surface and deep waters.

Thus, it appears that nitrogen fixation and denitrification tend to establish a self-regulating balance of nitrogen with other nutrients in eutrophic impoundments. Schindler (1977) pointed out that nitrogen-fixing blue-green algae develop in lakes that are fertilized with phosphate. This compensates for the relative deficiency of nitrogen. Denitrification provides a mechanism for eliminating excess accumulations of nitrogen. Neither nitrogen fixation nor denitrification are thought to be significant in unenriched lakes (Hutchinson, 1957).

Algae or plant growth in impoundments is related to the maximum nitrate-nitrogen concentrations that occur in early spring. Vollenweider (1971) indicates that lakes with very low productivity contain less than 0.2 mg/l and eutrophic lakes contain from 0.5 to 1.5 mg/l of nitrate-nitrogen. Because of the self-regulating feature just mentioned, the nitrate concentrations may be as much the effect of lake productivity as its cause.

Nitrate additions alone do not increase algal growth in the majority of lake waters. In samples of water from 49 widely distributed U.S. lakes, algal growth was limited by phosphate in 35, nitrate in 8, and other factors in 6 (Miller et al, 1974). The nitrate-limited lakes were mainly eutrophic lakes located in western states. In 29 lakes, addition of nitrate and phosphate increased growth more than addition of phosphate alone.

Little is known with certainty about the effects of nitrogen compounds on competition among plants in impoundments. Schindler (1977) showed that enrichment with nutrients other than nitrogen favors growth of blue-green algae. He therefore suggested that adding nitrate to keep a nutrient balance might favor the growth of more desirable green algae. Much research remains to be done in this area.

NITROGEN IN MARINE WATER QUALITY

Nitrogen is more frequently a limiting element for growth in marine environments than it is in streams or fresh water impoundments. We will consider briefly some nitrogen relations in the open ocean, in estuaries, and in near-shore areas.

Nitrate is the predominant form of nitrogen in deep ocean water. Concentrations generally range from 0.3 to 0.6 mg/l of nitrate-nitrogen (Sverdrup et al., 1942). The concentrations of nitrite- and ammonium-nitrogen rarely exceed 0.05 mg/l. The highest concentrations of nitrite are found only in a thin layer at depths from 100 to 400 meters. Concentrations of ammonium are more uniform through the water column.

In surface ocean water, biological uptake depletes the supply of inorganic nitrogen. Concentrations of particulate and dissolved organic nitrogen range widely from 0.0014 to 0.14 mg/l (Sverdrup et al, 1942). The higher concentrations are found where deep ocean water wells up to the surface, as along the Southern California coast or the western coasts of South America and Africa. Gradual incorporation of nitrogen into organisms, their death, and their sedimentation lowers the content of all forms of nitrogen in surface water of the open ocean. Recycling of organic nitrogen is necessary to support new growth in these areas. Ryther and Dunstan (1971) pointed out that a slower recycling rate for nitrogen than for phosphorus causes nitrogen to limit growth in much of the ocean.

Estuaries and near shore areas receive nutrients from land runoff. Concentrations of inorganic and organic nitrogen approach those in streams and impoundments in some areas. Up to 1 mg/l of nitrate-nitrogen is observed in parts of San Francisco Bay (Bain and Hoag, 1972). Along the south shore of Long Island, areas receiving runoff from duck farms developed dense blooms of algae (Ryther and Dunstan, 1971). Nitrogen was found to be the limiting nutrient for the algae in this case. Phosphorus and other nutrients were relatively more abundant in the runoff.

Nitrogen fixation and denitrification may be significant in marshy areas (Nixon et al, 1976). They are thought to be less important in the ocean in the short run (Ryther and Dunstan, 1971), although they may regulate the balance of nitrogen with other nutrients in the ocean over geologic time.

There is no evidence that nitrogen levels can change species composition in the marine environment. Frey (1977) found that nitrogen and phosphorus levels controlled the total growth of marine algae, but that micro-nutrients and vitamins had more effect on species composition. Carpenter and Guillard (1971) found that several species of marine algae developed strains adapted to different concentrations of nitrate-nitrogen.

SUMMARY AND CONCLUSIONS

The intended use of water determines the significance of various forms of nitrogen in water quality. Upper limit concentrations have been recommended for several forms of nitrogen in water used for public water supplies, freshwater aquatic life, marine aquatic life, and agricultural uses. The concentrations are lowest for cyanide, and increase in the order ammonium, nitrite, and nitrate. Nevertheless, nitrate is more often of concern in water quality than are the first three forms of nitrogen.

The recommended upper limit concentration for nitrate-nitrogen in public water supplies is 10 mg/l. The actual concentration in ground water exceeds this limit in some areas, and may be increasing. Because of the slow movement of ground water, increasing concentrations may take many years to correct.

Nitrate is depleted from surface waters by plant uptake, and may cause excessive growth of algae or other plants. This is seldom a problem in streams, but may become one in impoundments, estuaries, and near-shore ocean areas.

Much of the total nitrogen in surface water occurs in particulate or dissolved organic forms. Plant growth is often excessive when the total nitrogen concentration exceeds 0.5 mg/l if other nutrients are adequately supplied and growth conditions are favorable. Nitrogen supply limits growth in many ocean areas and in a few lakes, mainly eutrophic ones. Nitrogen fixation by blue-green algae tends to correct nitrogen deficiencies. Denitrification by bacteria tends to correct excesses. These features indicate that high nitrogen concentrations may be an effect, rather than a cause, of eutrophication.

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OVERVIEW OF NITROGEN IN IRRIGATED AGRICULTURE[1/]

R. S. Rauschkolb[2/]

Events of the recent past have been moving agriculture rapidly towards the point where it can no longer afford the luxury of being concerned with production of food and fiber alone. One of the principle reasons for the change in attitudes has been the Federal Water Pollution Control Act Amendment of 1972, commonly referred to as Public Law 92-500. Within this law there is a section dealing with areawide planning, Section 208. Under this section each state is to develop waste water management strategies which indicate methods for control or treatment of all point and non-point sources of pollution within an area. Specific outputs resulting from Section 208 areawide planning include a regulatory program to control or treat all point and non-point pollution sources, including in-place or accumulated pollution sources. This represents only one of the many outputs that are expected from area 208 planning, but it is one that seems to be the most important with respect to nitrogen management in irrigated agriculture.

Another factor, which will have an impact on the management of nitrogen, is the cost of energy. As fossil fuel becomes more costly and less available, the cost of fertilizer nitrogen will substantially increase which may result in the grower having to become more

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[1/] An abstract of this short paper is not provided because it is essentially an abstract to the introduction to a Report from R. S. Rauschkolb to the RSKERL, EPA (A. G. Hornsby), Ada, Oklahoma 74820, on Nitrogen Management in Irrigated Agriculture.

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efficient in the utilization of his fertilizer resources by virtue of cost. This does not appear at present to be a particularly strong motivating force for achieving efficiency because demand for food is going to increase as population increases, and there is no alternative to food production to meet the demand. Therefore, as cost of production increases, the increased cost will be passed on to the consumer. According to Stout[3/], the on-farm requirement for nitrogen to maintain a diet comparable to what the U.S. population is accustomed is about 82 kg/nitrogen/capita/yr. This is the amount of nitrogen required to provide the animal and plant protein we currently consume, and it also includes inefficiencies and various losses inherent in the agricultural food production system. Based on 1977 population this means about 18 million metric tons of farm-site nitrogen; about 50% of this nitrogen demand is met by manufactured commercial fertilizer nitrogen. The remainder of the nitrogen demand is made up by biological nitrogen fixation and recycling of nitrogen released from organic matter as well as natural chemical fixation.

Improvement in efficiencies and reduction of losses are certain to occur. It is felt these will be compensated for by increased population so that the farm-site nitrogen demand may remain fairly steady for a few years. However, the percentage of the farm-site nitrogen requirement met by the use of commercial fertilizers will in all likelihood increase.

Although this quantity of nitrogen seems quite large it is very small indeed in comparison to the total amount of nitrogen present in the earth and its atmosphere. The quantity is estimated by Delwiche (1970) to be on the order of 21×10^{15} metric tons of nitrogen. Using various estimates of the amount of nitrogen being fixed on a global basis, less than 1 millionth of a percent of the total nitrogen is being recycled. The amazing aspect of this is that in

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[3/] Stout, P. R. 1971. Agricultural requirements for nitrogen fertilizers in the USA. Unpublished report, University of California, Davis, California 95616.

spite of all the nitrogen that is present in the soil, water, air, plant and animal system, the general case in an agricultural system is for nitrogen to be deficient with respect to the amount of available nitrogen required for maximum plant growth. Consequently, there is the need to add nitrogen in order to produce the food and fiber needed.

The use of nitrogen fertilizers also has some beneficial effects with respect to utilization of our land and water resources. In a recent article on the environmental benefits of intensive crop production, Barrons (1971) shows a decrease in the number of acres used for plant production from approximately 365 million to about 290 million in a period from 1930 to 1970. During the same period, the population in the United States grew from approximately 120 to slightly more than 200 million. There was an increased food demand that was met by utilizing less acres for production of food and fiber. In the same article Barrons compared the number of acres required for the production of 17 crops in the 1938 to 1940 period to meet the production required for the 1968-1970 period for these same crops. Although other factors such as improved management, improved varieties, and pest control played an important role in the increased production, the widespread use of nitrogen fertilizer had a major impact on the increased production per unit area of land or per unit input of water. Consequently, it has substituted for increased development of water supplies and arable land. The number of acres saved, as indicated by Barrons, was approximately 292 million acres. In spite of the many benefits which can be ascribed to the use of nitrogen fertilizers there is ample evidence to indicate that inefficient utilization of our nitrogen fertilizer resources can lead to pollution of surface and ground water supplies. And, some of the pollution is attributable to the natural presence of nitrogen in vast quantities in our surroundings.

There has been considerable debate in recent years among scientists, environmentalists, and others in state and national institutions and agencies with respect to the best methods to employ for minimizing the nitrogen pollution potential coming from various sources. The debate has centered around two possible mechanisms for minimizing nitrogen pollution potential, and these are (1) treatment of the discharge waters or (2) control of inputs in order to minimize nitrogen output. For domestic and industrial waste water discharge where the discharge occurs at an identifiable point and where the volumes are relatively small, at least in comparison with the volume of water used in agriculture, it appears that treatment of waste water is the most appropriate method to minimize nitrogen pollution potential. In an agricultural system the situation is very different in that there are two types of discharges and volumes are much greater. One is an identifiable point source such as might occur in discharge of effluent from a drainage tile and the other system is a diffuse non-point discharge and is typified by water percolating below the root zone to an underground water table or by subsurface flow to return to the surface waters. Although the technology exists for treatment of the point source discharges, it is generally agreed that the volumes of water which must be handled make this an economical impracticable alternative. Because of the diffuse nature of percolating water and the virtual impossibility of collecting these waters for treatment, it is generally agreed, or at least conceded, that treatment of this non-point sources of nitrogen pollution is not a viable alternative.

Monitoring of nitrogen concentration in percolating waters has been suggested as one technique for evaluating nitrogen pollution potential of these percolating waters. The idea being that if unsatisfactory levels were detected then corrections could be made in the amount of nitrogen input. However, one of the problems inherent in using this technique is the extreme variability which is known to occur with nitrogen concentration in soils. In a recent study in California, funded by the Research Applied to National Needs Division of the National Science Foundation, it was found by Rible et al (1976) that in order to be within 20% of the mean nitrate-nitrogen concentration when there were 9 samples taken per hole, it would require anywhere from 5 to 20 holes per site to reach that level of precision. They found the number of holes that were required varied depending upon irrigation method, crop type and soil texture. Even if one could afford the cost incurred to reach the level of precision indicated, that level of precision is not satisfactory as a method for determing whether excessive nitrogen inputs have been made or not. In addition, that level of precision is thought by some to be no better than what might be achieved through nitrogen management and water management to control nitrogen pollution potential. Furthermore, it is even expected that some control measures will result in a reduction of cost rather than an increase in cost.

So, it appears that the least costly, most effective method for minimizing nitrogen pollution potential in irrigated agriculture is to employ management techniques which result in the greatest crop output for each unit of input of nitrogen and water. In addition to being an effective means of minimizing nitrogen pollution potential, this technique also allows the use of a simple method to monitor the effectiveness of control measures. By use of figures that are easily obtainable from existing farm records it is possible to determine the crop output per unit of nitrogen input. However, the variations which occur in an agricultural system and in the management variables that may be employed in an agricultural system will determine the peak efficiency that may be attained. Using this approach, peak operational efficiency can be the goal which will lead to greater nitrogen use efficiency with a concomittant reduction in the nitrogen pollution potential.

An understanding of the factors which are operating in the agricultural plant production system is essential in order to understand how various techniques of nitrogen and water management may be employed on a site specific basis to reduce the nitrogen pollution potential. This approach takes into account the variation that occurs within the agricultural system and applies management technology to achieve a minimization or elimination of unwanted entry of nitrogen into surface and ground water resources. There are the independent system variables such as soil, crop, irrigation methods, nitrogen cycle, and environment (SCINE) which defines the system within which management techniques can be employed. Although it is recognized that not all these major factors delineating the system are entirely independent of one another they are sufficiently independent of one another as to warrant their separate consideration in attempting to learn how to adapt management alternatives to mitigate unwanted effects.

Another aspect of the system is the dependent management variables which can be employed within a given set of conditions to increase nitrogen use efficiency. These management variables are placement, equipment, rate, source, irrigation management and energy (organic matter) which allows the assignment of an acronym PER SITE. This assists in recognition of those management variables which are functioning within a given set of conditions for the agricultural system and at the same time connotes the importance of considering the site specific factors which determines how each of the management techniques will be employed. A feature of the management techniques and indeed the problem that makes their application difficult is that they are so greatly interdependent. Learning how to utilize combinations of management variables to have an impact on reducing nitrogen pollution potential is the challenge facing those charged with mitigating possible pollution effects.

Methods of implementing on-farm control measures for nitrogen management in irrigated agriculture will require field personnel with an understanding of soil, water and plant relationships who can guide the farmer in the application of control measures which provide flexibility, ease of implementation and are adaptable to a wide range of conditions and managerial abilities. Agriculture does not exist independently from all other segments of society. Consequently, it shares concerns of others with regard to conservation of energy, land and water resources as well as the impact of man's activities on the environment. In addition, it is in agriculture's best interest to maintain an environment compatible with the needs of others since such an environment is also essential to the continued production of food and fiber.

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SOURCES OF NITROGEN FOR CROP UTILIZATION

L. S. Murphy[1]

ABSTRACT

Nitrogen fertilization of crops is essential for maximum food production. Nitrogen from organic matter in the soil declines with continued cultivation to the point that supplemental nitrogen applications are necessary. Supplemental sources of nitrogen for crop production include symbiotically fixed nitrogen from legumes, non-symbiotically fixed nitrogen from free-living organisms in the soil, inorganic nitrogen from lightning discharges and industrial emissions, industrially fixed nitrogen and nitrogen from various waste products.

All sources of nitrogen for plants with the exception of symbiotically and non-symbiotically fixed nitrogen undergo the same chemistry in the soil from application until plant use. Under aerobic conditions, all nitrogen applied to the soil will eventually reach the nitrate form. Plants are therefore unable to distinguish the original source of nitrogen.

Symbiotically fixed nitrogen supplies important amounts of this element to legumes and crops which follow the legumes. Industrially fixed nitrogen plays a major role in supplying nitrogen to crops in most areas of the world. Characteristics of manufactured nitrogen products differ more than those of plant and animal residues. Those characteristics and their effects on utilization of these nitrogen sources are discussed in detail in this review. Also discussed are practices intended to maximize the efficiency of applied nitrogen from any source.

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Use of animal wastes is discussed with emphasis on determining application rates by nutrient needs.

Recommendations for nitrogen applications in either dryland or irrigated agriculture are based on crop need as determined by agricultural research. Soil analysis is an important tool in determination of needs. Use of present and future recommendations will both maximize production of food and maintain environmental quality.

INTRODUCTION

Most of the nitrogen in the Earth and its atmosphere is the inert gas N_2 which represents about 78% of the atmosphere. In that form, however, nitrogen cannot be utilized directly by higher plants. This chapter will serve to outline the processes involving conversion of elemental nitrogen into fixed forms which can be utilized by plants but will focus on properties and use of nitrogen sources. Fixation by both the activities of symbiotic bacteria such as Rhizobia and also by the techniques which have been devised industrially to convert elemental nitrogen into fixed forms will be considered.

The characteristics of nitrogen sources including their agronomic properties will be considered.

FORMS OF SOIL NITROGEN

Most of the nitrogen which exists in the soil is present in organic matter and occurs in a very large number of compounds including many of the nitrogen-containing materials normally found in plants. If we analyzed the soil for its organic nitrogen compounds, we would find some proteins, amino acids, amides, amines, aklaloids, nucleic acids, nucleotides, and others. The fact which we should keep in mind concerning all these organic compounds, however, is that the nitrogen contained in them is all essentially the same chemical form. Nitrogen in these organic compounds exists in the form of amino (NH_2^-) groups which represent the same oxidation state (-3) as represented by ammonia (NH_3) and ammonium (NH_4^+).

SYMBIOTIC NITROGEN FIXATION

An important contribution to nitrogen nutrition of all plants is made by certain species, principally the legumes which enter into a symbiotic relationship with certain soil bacteria. In this relationship, the bacteria inhabit nodules on host plant roots and are provided with carbohydrates by the host plant. The bacteria subsequently convert elemental nitrogen into a form usable by the host plant. The contribution of this form of nitrogen to plant nutrition in general is substantial. Quantities of nitrogen fixed in any symbiotic relationship between bacteria such as _Rhizobia_ and a legume such as alfalfa vary with the strain of the organism, the condition of the host plant and various environmental conditions. Nitrogen fixation by this process has been estimated as high as 500 kg nitrogen/ha by a crop of clover in New Zealand. However, climatic conditions and soil conditions must be optimum for such large amounts. Some indication of the amount of nitrogen fixed by various legumes is presented in Table 1.

The type of bacteria which produces nitrogen in the nodules on roots of one crop may not be adapted to a second host plant. For instance, the inoculum which is normally applied to soybeans prior to seeding to insure adequate numbers of _Rhizobia_ for nitrogen fixation is not the same strain that would be utilized in inoculating alfalfa.

Environmental conditions favoring nitrogen fixation symbiotically include a relatively high soil pH (7.0 or higher), adequate soil moisture, a good supply of available calcium, warm temperatures, and a low supply of available inorganic soil nitrogen. Symbiotic nitrogen fixing bacteria have a high requirement for calcium, thus the relationship of both pH and available calcium to their activity.

Temperature affects the activity of any biological process, and thus, cooling the soil temperature to near freezing reduces nitrogen fixation to near zero. The temperature would also affect the host plant. Inadequate numbers of bacteria in the soil obviously would reduce fixation. These organisms are normally added to the seed of legumes at least when the production of such legumes is initiated in a given field.

Referring back to inorganic nitrogen supplies, researchers have reported that applications of available nitrogen prior to legume seeding reduce the number of nodules and reduce the amount of nitrogen fixed by the host plant-bacteria relationship. Apparently, the activity of the organisms is sensitive to the presence of such forms of nitrogen as ammonium and nitrate. An indication of the effects of nitrogen on nitrogen fixation under controlled conditions and on number of nodules on soybean plants is presented in Tables 2 and 3.

Use of symbiotically fixed nitrogen by other than the host plant involves the residual effects of legume production. Nitrogen contained in the roots and straw of the preceeding crop becomes available following mineralization. Voss (1978) has provided some estimates of the contribution of a legume to a following crop (Table 4). Note that the contribution is based on the amount of legume in the stand of plants preceeding the crop in question. Raney and Sharplaz (1975) reported yield increases of irrigated corn as high as 3300 kg/ha in the third crop following two years of alfalfa (Table 5). Legume sources of nitrogen should be taken into account in the following crop and proper adjustments made in the nitrogen to be applied to that crop. This adjustment is important with regard to costs of production, efficient use of nitrogen and control of leaching of nitrate-nitrogen toward shallow aquifers. The contribution from a legume to the following crops will decline with time. In the Raney and Sharplaz data,

Table 1. Estimated fixation of nitrogen by legumes.*

Legume	Nitrogen Fixed Kg/ha	Legume	Nitrogen Fixed Kg/ha
Alfalfa	220	Lespedezas (annual)	95
Ladino clover	200	Vetch	90
Sweet clover	130	Peas	80
Red clover	125	Soybeans	110
Kudzu	120	Winter peas	60
White clover	115	Peanuts	47
Cowpeas	100	Edible beans	45

*Compiled from Alexander (1961) and Tisdale and Nelson (1975).

Table 2. Effect of ammonium on nitrogen fixation in gravel culture.*

Ammonium nitrogen added	Sources of Plant Nitrogen		Portion of nitrogen from air
	Fertilizer	Air	
mg/pot	mg	mg	%
	Soybeans		
0	0	1639	100
80	68	1692	95
320	252	2243	89
560	464	2185	82
800	648	2423	79
	Ladino clover		
0	0	188	100
80	63	234	75
320	282	159	35
560	527	98	15
800	609	82	12

*Allos and Bartholomew (1959).

Table 3. Effects of nitrogen carriers and time of nitrogen application on the nodulation of dryland soybeans. All carriers were added at a rate of 168 kg nitrogen/ha.

Nitrogen Carrier	Time of Application	Nodules/ 5 Plants	Mean Values	Nodules/ 5 Plants
No nitrogen	---	445	Ammonia	171
Ammonia	Pre-plant	144	Urea	294
Ammonia	Split	281	Nitrogen solution	251
Ammonia	Sidedress	391	Calcium nitrate	294
Urea	Pre-plant	141	SCU-30	226
Urea	Split	253	$\text{LSD}_{.05}$ for Nitrogen Carrier	NS
Urea	Sidedress	488		
Nitrogen Solution	Pre-plant	125	Pre-plant	134
Nitrogen Solution	Split	182	Split	251
Nitrogen Solution	Sidedress	447	Sidedress	416
Calcium nitrate	Pre-plant	142	$\text{LSD}_{.05}$ for Time of Application	56
Calcium nitrate	Split	295		
Calcium nitrate	Sidedress	445		
SCU-30[2]	Pre-plant	118		
SCU-30	Split	247		
SCU-30	Sidedress	313		
$\text{LSD}_{.05}$ Treatment		122		

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[1] Meyer et al (1974).

[2] Sulfur-coated urea.

Table 4. Corn yields from two crop-rotation fertility experiments-Iowa. Data are in kg/ha.

Cropping system	Nitrogen added, kg/ha					
	0	40	67	90	134	202
	Clarion-Webster Research Center, 1972-76					
Continuous corn	2885	--	3763	--	6209	7275
Corn after soybeans	5896	--	6586	--	7526	7589
Corn after 1 yr. meadow	8091	--	7777	--	8091	8216
Corn after 2 yr. meadow	8655	--	7840	--	7150	8091
	Northwest Research Center, 1957-76					
Continuous corn	4390	5833	--	6397	6648	--
Corn after soybeans	5832	6523	--	6836	7025	--
Corn after 1 yr. meadow	6460	6648	--	6962	--	--
Corn after 2 yr. meadow	7025	6899	--	6899	--	--

Estimated nitrogen contribution of the legume crop to the following corn crop.

First-year corn following:	Kg nitrogen/ha contribution from legume
50-100% legume meadow	157
20-50% legume meadow	112
0-20% legume meadow	22
Legume green manure	112
Soybeans for grain	1 Kg nitrogen/60 Kg soybeans

Voss (1978)

Table 5. Effects of nitrogen, phosphorus, and residual nitrogen from alfalfa on irrigated corn.*

Nitrogen	Phosphorus	Corn yield no alfalfa	Corn yield third crop after 2 years alfalfa	Yield differences due to alfalfa effects	
				1975	1974
kg/ha					
0	0	2885	3981	1129	2070
90	0	7652	9596	1944	1693
90	49	6460	9784	3324	2822
179	49	10349	10035	-313	1505
267	0	11415	13171	1756	-63
267	49	12293	12544	251	-125
Mean		8530	9847	1317	1254
LSD .05		1631	2822	----	----

*Raney et al (1975)

Table 6. Nonsymbiotic nitrogen fixation under variable conditions (Delwiche et al, 1956).

Treatment	Soil	Fixed nitrogen present as		Total nitrogen fixed
		Ammonia	Nitrate	
		meq/100 g soil		
10 g. Yolo soil + 780 mg. sucrose	Davis	0.055	0.028	0.083
Growing lawn, photosynthesizing	Davis	--	0.0004	0.0004
	Berkeley	0.004	0.001	0.005
Inverted lawn, decaying	Davis	0.18	0.26	0.44
	Berkeley	0.087	0.047	0.134
Soil with grass removed	Davis	0.001	0.007	0.008
	Berkeley	0.001	0.002	0.003

however, the contribution to the third corn crop was just about the same as that contributed by the alfalfa to the second corn crop.

NON-SYMBIOTIC NITROGEN FIXATION

Fixation of elemental nitrogen by bacteria other than those inhabiting the nodules of legumes and other plants has recently come under scrutiny. Nitrogen fixation by certain organisms which exist free of any host plant relationship has been indicated by studies in soil microbiology. The organisms which are involved in such nitrogen fixation include blue-green algae and certain free living bacteria such as *Azotobacter*, *Clostridium*, and *Rhodospirillum*. *Rhodospirillum* is a photosynthetic organism while *Clostridium* and *Azotobacter* are saprophytic organisms of anaerobic and aerobic characteristics, respectively.

Photosynthetic organisms, including blue-green algae and *Rhodospirillum*, are thought to make only minor contributions to the nitrogen in upland agricultural soils because of the requirement of sunlight in the physiological activity of these organisms. They may play a more major role, however, in soil which is wet for extended periods such as the culture of rice.

Other organisms obtain their energy from the oxidation of organic carbon (organic matter) and may be able to contribute more nitrogen to the environment than the photosynthetic organisms. Microbiologists have estimated that nitrogen fixed by some of these organisms may range from 20-30 kg/ha annually but most researchers feel that the value is much lower, around 7 kg/ha. Some indication of the amount of nitrogen which has been fixed by free living organisms in a given soil is presented in Table 6 (Delwiche et al, 1956).

ATMOSPHERIC SOURCES OF FIXED INORGANIC NITROGEN

Lightning results in the conversion of some elemental nitrogen (N_2) to oxides of nitrogen such as nitric oxide (NO) and nitrogen dioxide (NO_2). Lightning produces extremely high air temperatures which in the presence of oxygen and elemental nitrogen result in the production of nitric oxide. This gas combined with oxygen to produce nitrogen dioxide (NO_2) which is absorbed in water producing nitric acid. Nitrogen oxides from industrial sources and automobile exhaust also provide a source of nitric acid in rainfall.

Another form of nitrogen in the atmosphere which has become increasingly common with industrialization is ammonia gas. This gas is soluble in water and is readily washed out of the atmosphere with rainfall. It is also absorbed directly from the air by surface water bodies and by soil colloids. Studies in industrialized areas have indicated that 60 to 70 kg nitrogen/ha could be supplied as ammonia from industrial contamination of the atmosphere.

The contribution of fixed atmospheric nitrogen to the soil from all sources is generally less than the amounts quoted above. Considering the low industrialization of much of the area of the U.S. and the world, contributions of these sources to soil nitrogen supplies is probably less than 10 kg/ha/year on the average.

COMMERCIAL NITROGEN FERTILIZERS

Anhydrous Ammonia Production

The production of most nitrogen fertilizers in the world is dependent upon the synthesis of anhydrous ammonia (NH_3). Commercial production of this compound is based on the discoveries of two German chemists, Fritz Haber and Carl Bosch, in 1908. Their work with equilibrium chemistry and the subsequent development of a suitable catalyst for the combination of nitrogen and hydrogen under pressure led to development of the process used throughout the world. The first small ammonia synthesis unit was placed in operation in 1911. Properties of anhydrous ammonia are presented in Table 7.

Table 7. Properties of anhydrous ammonia.*

Color	Colorless
Odor	Pungent - Sharp
Molecular weight	17.03
Weight per gallon at 60F	5.14 lb
Weight per cubic foot at 60F	38.45 lb
Boiling point at 760 mm Hg	-33.35C (-28F)
Freezing point at 760 mm Hg	-77.7C (-107.9F)
Heat of fusion	79.4 cal/g (143BTU/lb)
Heat of vaporization at 760 mm Hg	327.4 cal/g (589.3 BTU/lb)
Critical temperature	132.4C (270.3F)
Critical pressure (absolute)	111.5 atm (1,639 lb/in^2)
Critical density	0.235 g/ml (14.67 lb/ft^3)
Dielectric constant at 760 mm Hg	
Vapor at OC, 1×10^6 cycles/sec	1.0072
Liquid at -34C, 4×10^8 cycles/sec	22.0
Solid at -90C, 4×10^8 cycles/sec	44.01
Electrical conductivity of liquid at freezing point	13×10^{-8} mho/cm
Viscosity	
Liquid at -33.5C	0.266 centipoises
Vapor at -78.5C	0.00672 centipoises
0.0C	0.00926 centipoises
20.0C	0.01080 centipoises
100.0C	0.01303 centipoises
Solubility in water	Very soluble
Solubility in alcohol	Soluble
Heat of formation gas (26C, 1 atm)	1,160 BTU/lb (644 cal/g)
Decomposition temperature	871.1-872.2C (1,600-1,800F)

*McVickar et al, 1966.

Anhydrous ammonia production in the United States alone totaled 15.36 million metric tons in 1977. In the United States, Australia and certain parts of western Europe, anhydrous ammonia is utilized for direct soil application. In most other countries of the world, anhydrous ammonia is utilized entirely for production of other fertilizers and other industrial uses, but little is consumed by direct application. Direct application of ammonia in the United States accounts for about 40% of the nitrogen applications.

Direct Application of Anhydrous Ammonia

Application in Irrigation Water.

The first use of anhydrous ammonia for direct application as a fertilizer involved injection of ammonia into irrigation water in California (Waynick, 1934, Rosenstein, 1936). These early efforts demonstrated that ammonia was a feasible source of nitrogen but also revealed that problems can occur when ammonia is injected into water containing large quantities of dissolved calcium and magnesium salts (Warnock, 1966).

When ammonia is injected into irrigation water, an equilibrium is set up which generates hydroxyl (OH^-) ions along with ammonium (NH_4^+) ions.

$$\underset{\text{ammonia}}{NH_3} + H_2O \rightleftharpoons \underset{\text{ammonium}}{NH_4^+} + \underset{\text{hydroxyl}}{OH^-}$$

The hydroxyl ions increase the pH of the water causing the solubility of dissolved salts to decline, particularly calcium salts. Subsequently, these salts precipitate, nozzles are clogged in sprinkler systems, pipe weight increases substantially in the case of surface systems and water flow is restricted.

Ammonia injection into ditch or siphon tube irrigation systems is less troublesome. Siphon tubes can become encrusted with the precipitated calcium salts but are more easily cleaned. If the precipitation occurs in the irrigation ditch, no particular problems are encountered.

Some relief from problems of ammonia injection into hard water can be obtained by using an inhibitor such as sodium hexametaphosphate which tends to sequester the calcium reducing precipitation. Agricultural engineers at the University of Nebraska have published guidelines for precipitation inhibitor use (Mulliner, 1974) based on amount of ammonia being applied and the quality of the water. That information is summarized in Table 8.

Table 8. Suggested anhydrous ammonia and calcium precipitation inhibitor rates (adapted from Mulliner, 1974).

Irrigation water hardness	Maximum ammonia application per 4000 lH_2O/min	Inhibitor required[1/] per 4000 lH_2O/min
ppm	kg/hr	g/hr
35-120	18	120
120-170	18	180
170-425	18	300
425-850	13	360

[1/] Sodium hexametaphosphate, "Calgon".

Another major problem in applying anhydrous ammonia or free ammonia-containing solutions through sprinkler irrigation systems is the fact that volatilization losses of ammonia will occur from the time that the ammonia-water mixture leaves the sprinkler until the ammonia reaches the soil surface. Admittedly, some of the nitrogen is present as ammonium ions but much more is in the ammonia form. Subsequently, as water vaporizes, ammonia is also lost to the atmosphere as a gas.

There has been much controversy over the actual amount of ammonia lost by such a process. Most of the work in that area was carried out in the mid-1950's. Scott (1956) suggested that the net loss might range from 5 to 40%. Jackson and Chang (1947) estimated losses to range as high as 58% while Henderson et al (1955) presented data showing losses as high as 60%. Recently, University of Nebraska research has corroborated these findings. Conditions governing such losses include initial pH of the water, temperature at the time of application, wind velocity, and concentrations of ammonia in the water.

Application of ammonia or any other nitrogen source in irrigation water (fertigation) is appealing to the irrigation farmer because of possible economies in application costs and improved management of the nutrient supply. Applications close to the time of plant use may avoid nitrogen losses by leaching and denitrification. Soil compaction may be reduced by elimination of a trip over the field.

Ammonia or any nitrogen source applied in irrigation water is distributed in about the same manner as the water. Poor sprinkler patterns are not too much of a problem with ammonia application because of the already overwhelming problem of volatilization losses. Application through gated pipe or siphon tube systems must consider the fact that more water percolates into the soil at the head of the run than at the tail. Similarly, more ammonia would be expected to be absorbed by colloids at the head of the run. To offset this problem, ammonia is frequently withheld from the water during the first part of the irrigation avoiding part of the accumulation at the head of the run.

Fischbach (1964) recommends that nitrogen injection begin as soon as the water stream is started down the furrow and continue until the water and fertilizer reach the end of the run. Fertilizer application should then be shut off and water application continued until the proper amount of water has been applied.

Leavitt (1966) commented on his early experience with ammonia application in irrigation water for contour irrigation of rice and noted that severe problems were encountered due to absorption of all of the ammonia by soil and organic matter during the stages of migration of water through the contour system. All ammonia was absorbed in the first 2-3 kilometers of travel.

Conventional Soil Application. Development of ammonia application equipment began soon after the initial work with ammonia application in irrigation water. Interestingly, the original designs are not radically different than the types of "shank" applicators in use today. Many designs of ammonia applicators exist, consideration of which is far beyond the scope of this paper. Basically, ammonia equipment consists of a rigid or flexible shank set to penetrate the soil to a predetermined depth. A tube is attached to each shank which opens 2-5-cm above the point of the shank. Ammonia is delivered to this metal tube by flexible plastic lines from the device controlling the flow of ammonia from the tank.

Spacings between points of ammonia release in the soil vary greatly between different cropping systems. Basically, uniform plant growth is the goal of the equipment design but considerations must be given to practical aspects of draft and equipment wear. Obviously, since ammonia has an appreciable vapor pressure, all applicator designs for both anhydrous ammonia and aqua ammonia must provide for sealing ammonia in the soil by dragging soil into the knife opening. Improper sealing, insufficient depth of application, excessively wide spacing between release points and high rates of ammonia application can all contribute to volatilization losses.

The soil retention pattern produced by ammonia injection is an important function of ammonia use. Early efforts in developing commercial ammonia application involved distribution pattern studies (Leavitt, 1966). The ammonia retention zone was noted to be about 15 cm in diameter with elongation to the top because of the broken soil zone produced by passage of the ammonia knife. Continuing research into the 1970's has corroborated those early findings. Distribution patterns vary with soil cation-exchange capacity, moisture content, magnitude of application and spacing of ammonia knives. That retention zone should not come into contact with the soil surface otherwise volatization losses will occur.

New Developments in Ammonia Application. Achron et al (1977) recently surveyed the industry for new developments and identified a growing area of interest in ammonia application, namely adaptation of tillage equipment for ammonia application during tillage operations. This practice is not totally new but the concept has increased merit today because it poses the possibility of eliminating or combining operations with subsequent savings in time and energy. The imagination of ammonia users has led to development of ammonia application capabilities for field cultivators, moldboard plows, tandem and offset discs, undercutting equipment, rod weeders, bedding equipment and a host of others.

Recently, a system designed to aid in adaptation of ammonia application by tillage implements was introduced. This system known as Cold-Flo ammonia (Achorn, 1977) allows shallower ammonia application by reducing the vapor pressure of the ammonia by allowing the ammonia to boil, act as its own refrigerant thus cooling itself to -33C. Vapor pressure of the liquid ammonia is essentially zero. Some of the ammonia must be injected into the soil as a vapor, however. At 25C, about 15% of the ammonia is converted into vapor in the boiling process and obviously must not be lost. Still, this vapor is under relatively low pressure and does not have to be injected so deeply. This process seems to be particularly well adapted to areas where ammonia application under wet soil conditions may be necessary. If the lower soil profile is wet but the surface is dry enough for equipment operation, sealing could be a problem. The shallower depth of application of Cold-Flo ammonia then could allow producers to use the lower cost (per unit of nitrogen) source.

Time of Ammonia Application. This point alone could be the subject of an entire chapter. Without all the discussion pro and con of fall pre-plant ammonia versus spring pre-plant versus sidedress nitrogen for row crops, let's remember a couple of things about ammonia application. The closer the application to the time of crop need, the greater the efficiency possibly may be. Less loss by leaching and/or denitrification may be a result of careful selection of application time. On the other hand, applications very close to the time of seeding (pre-plant) create a greater chance for germination damage from ammonia.

When ammonia is injected into the soil, it immediately begins to react with water. Subsequently, the ammonia and any seedlings in the vicinity compete for available soil moisture. Plants close to the ammonia retention zone experience high osmotic pressure of the soil solution in the vicinity of the retention zone and also may be affected by ammonia toxicity directly. Good rules of thumb to help avoid the problem of germination damage are to (1) apply the ammonia as far in advance as possible, (2) to use as narrow release points as feasible, (3) to apply as deeply as practical, and (4) to apply the ammonia at an angle to the direction in which the rows will be planted. Suspected ammonia germination damage can be diagnosed in several ways including pH of the soil surrounding the damaged seedling (high pH probably means high ammonium concentration), and ammonium concentration in the soil samples surrounding the damaged seeding. Colliver and Welch (1970) studied the effect of rate, depth, and time of ammonia application before corn planting on the germination and early growth. Their work clearly indicated that as depth of application increased from 10 to 17.5-cm, effects on stand were essentially nil at rates of 112 and 224 kg nitrogen/ha. Applications immediately prior to planting were still acceptable as long as the seed and the ammonia retention zones were separated.

Agronomic Comparisons of Ammonia and other Nitrogen Sources.

Since the development of ammonia as a nitrogen source for direct application, comparisons of the performance of this material and other nitrogen sources have been carried out. Smith (1966) discussed comparisons of materials and concluded that ammonia is an effective source of nitrogen for a variety of crops, including row crops, small grains and forages. Smith noted in his discussion of ammonia application that time of application and rate of application, each a factor in determining final agronomic results, cannot be arbitrarily specified without consideration of many soil and climatic factors.

Results of comparisons of nitrogen sources for any crop will vary with years but usual differences between carriers are small when several years' means are considered. An example of the variation that can result between such comparisons is indicated in Table 9.

Table 9. Nitrogen sources for irrigated corn (Gallagher et al, 1975).

Nitrogen Carrier 168 kg nitrogen/ ha/yr	Corn Yield, kg/ha					
	1971	1972	1973	1974	1975	5-year Mean
Control	5268	7275	5519	2571	2007	4516
Ammonia	6836	8718	7903	7526	9031	8078
Urea	7338	7714	8843	7401	6648	7589
Sulfur-coated urea	6836	7526	8906	8028	6335	7526
LSD_{05}	501	1066	752	NS	564	696

Ammonium Nitrate

Ammonium nitrate (NH_4NO_3) was the first solid nitrogen product that was produced on a major industrial scale. After World War II, plants which had been designed to produce ammonium nitrate for munitions were converted to fertilizer production. Plants today involve a process for conversion of ammonia into nitric acid and then react the nitric acid with ammonia to produce ammonium nitrate. The properties of solid ammonium nitrate are presented in Table 10.

Table 10. Properties of ammonium nitrate.

Color	White crystalline
Molecular weight	80.04
Density (24C)	1.725
Melting point	169.6C
Boiling points	210C
Solubility	118 g/100 ml at 0C 871 g/100 ml at 100C

Table 10. Continued

Nitrogen, % (pure)	35
Nitrogen, % (fertilizer grade)	33.5-34

The U.S.A. ammonium nitrate production was 4.18 million metric tons in 1977.

Use of Ammonium Nitrate. Probably fewer agronomic precautions are necessary with the use of ammonium nitrate than with any of the other materials discussed. Direct seed placement of ammonium nitrate in either irrigated or dryland agriculture can pose problems of seed germination through salt effects. The amount of ammonium nitrate which can be placed in direct seed contact varies with crop, soil type, water supply and the amounts of other nutrients placed in the same soil zone. As a general rule, greater than 20 kg of nitrogen/ha in direct seed contact as ammonium nitrate might be expected to produce problems. These problems relate strictly to water availability to the seedlings so irrigated conditions might reduce the problem. It is not peculiar to ammonium nitrate, other soluble salts can create the same problem.

Soil chemistry of ammonium nitrate is straight forward and has been covered adequately in another chapter of these proceedings (Broadbent, 1978). Oxidation of the ammonium portion of ammonium nitrate by soil bacteria does generate soil acidity but an identical effect is produced by the oxidation of ammonium ions from any source including plant residues and animal manures. Tisdale and Nelson (1975) have summarized the acidity-basicity of nitrogen fertilizers and point out that nitrification of one kg of ammonium produces sufficient acidity to require 1.8 kilos of pure calcium carbonate for neutralization. Half of the nitrogen in ammonium nitrate is already in the nitrate form and does not undergo nitrification. Liming becomes increasingly important as ammoniacal fertilizers (those supplying ammonium nitrogen) are used for extended periods.

The effect of ammonium fertilizer use on soil pH is a function of the soil's content of free calcium carbonate, the cation-exchange capacity of the soil and the soil's mineralogy. This discussion is not intended to focus attention primarily on ammonium nitrate, rather the problem is a universal one and net effects of continued use of ammoniacal fertilizers can only be monitored by soil testing. European farmers and fertilizer manufacturers have attempted to modify this effect by producing several materials known as ammonium nitrate of lime (ANL) or calnitro. These materials are merely mixtures of ammonium nitrate and calcium carbonate. Liming is then occuring when nitrogen is applied. Depending on the formulation (ranging from 16-20% nitrogen) the net effect of continued use may be either to maintain the soil pH or to increase it slightly. The obvious disadvantage of such materials is the low nitrogen content. Only very small quantities of such materials are used in the U.S.

Surface applications of ammonium nitrate have generally not been as subject to volatilization losses of ammonia as similar applications of urea. Rolston (1978) has reviewed this subject.

Ammonium nitrate, while having excellent agronomic capabilities does have some specific characteristics which can make the material hazardous. Contamination with organic materials such as carbon black, diesel fuel or ground plant material provides a combination which is easily oxidized in the presence of ammonium nitrate. Ammonium nitrate is classified as hazardous product due to the fact that it is a strong oxidizing agent. The presence of carbon, high temperature and pressures in the vicinity of 36 kg/cm^2 can cause ammonium nitrate detonation. The strong oxidizing tendency of ammonium nitrate is reason enough to protect this material in storage from contamination with carbonaceous material.

For years ammonium nitrate has reigned supreme among nitrogen carriers in the world for direct application. Over the last several years, however, a decline in ammonium nitrate usage has been noted with the rapid rise in production of urea. Ammonium nitrate production today is subjected to severe environmental restraints, both from the release of nitrogen oxides (NO and NO_2) to the atmosphere and also from the dust produced in the prilling and coating process. New ammonium nitrate plant construction in the United States is rare and future use of this material for direct application will probably decline. This should not be interpreted as an indication that ammonium nitrate will not be manufactured because it is a necessary component of nitrogen solutions.

Urea

The existence of urea ($CO(NH_2)_2$) in nature has been recognized for over 200 years but commercial synthesis of this material was not achieved until the 20th century. Synthesis of urea for both fertilizers and an animal feed additive involves the use of anhydrous ammonia and carbon dioxide generated in the production of anhydrous ammonia. Characteristics of solid urea are indicated in Table 11.

Table 11. Properties of urea.

Color	White
Molecular weight	60.06
Density crystalline	1.32 g/cc
Density, fertilizer	42 lb/cu. ft.
Solubility	78 g/100 ml at 5C
Melting point	132.7C
Boiling point	Decomposes at atmospheric pressure
Nitrogen, % (pure)	46.67
Nitrogen, % (fertilizer grade)	45-46

United States urea production in 1977 amounted to 3.09 million metric tons.

Use of Urea. Urea reactions in the soil have been described earlier in these proceedings. Some of those soil reactions relate to the efficiency of urea as a nitrogen source for dryland or irrigated crops. One of the most persistent problems with use of urea has been that of ammonia volatilization following hydrolysis. This problem, has been most troublesome on high pH soils or on soils where a large amount of crop residue limits urea contact with the soil surface. Research has clearly demonstrated that incorporation of urea into the soil will reduce problems of ammonia volatilization to a workable level (Rolston, 1978).

Surface applications of urea are not always conducive to problems, however. Dibb and Welch (1974) noted that aerial top dressing of corn with prilled urea in a wet spring when ground application was not possible proved feasible with only slight leaf burn from material trapped in the whorl. Comparisons of urea with many other nitrogen sources have shown that this material is a very excellent nitrogen source (Table 18). Engelstad and Hauck (1974) have recently summarized the productive capabilities of urea. They note, as do others, the emerging importance of urea as a solid nitrogen source, frequently at the expense of ammonium nitrate. Solubility of urea makes possible the use of this material as a foliar nitrogen source for many crops. Less leaf burn from foliar applications of urea as compared to ammonium nitrate results from lower ammonium ion concentrations on the leaf surface. Foliar applications of urea have been used successfully with many crops particularly high value crops such as citrus where repeated sprayings are a part of normal production practices. Inclusion of urea in sprays does allow for close control of nitrogen concentrations in plant tissue. Late growing season applications of urea solutions have been used to increase the protein content of wheat (Finney et al, 1957). Apparently, such applications at flowering are readily absorbed and translocated into the filling grain as functional protein.

Urea-ammonium nitrate combinations have become increasingly popular in formulation of nitrogen solutions. These two compounds exert a synergistic effect on each other's solubility producing much higher nitrogen concentrations (28-30% nitrogen) than can be achieved by a urea-only or ammonium nitrate only solutions (approximately 18-20% nitrogen). Urea-ammonium nitrate (UAN) solutions are not commonly used for foliar applications because of leaf burn from the ammonium in the ammonium nitrate.

Seedling Damage. Urea in direct seed contact has caused problems with seedling survival in several crops, particularly wheat, corn and grain sorghum. Urea hydrolysis produces ammonium ions or ammonia molecules depending upon the soil conditions. Characteristics of the soil zone where urea hydrolyzes include high pH from the hydroxyl ions (OH^-) produced and possibly free ammonia resulting from the high pH. Free ammonia is toxic, apparently interfering with electron transport in the plant. Also, ammonia has a high affinity for water and in sufficient concentrations exerts a desiccation effect on seedlings.

Susceptibility to these problems varies with amount of urea applied, the clay-organic matter content of the soil, pH and variety. Applications of urea as low as 10 kg nitrogen/ha in direct seed contact have caused extreme problems in grain sorghum emergence. Like other phenomena, this problem is not totally predictable. However, it is of high enough probability to suggest that urea should not be used in a starter fertilizer that will come into direct seed contact. Placement of urea away from the seed eliminates this problem.

Biuret. Biuret, a condensation product of urea, was identified many years ago as a toxic agent. Brage (1960) studied the effects of biuret on winter wheat in South Dakota and noted a substantial reduction in stand when biuret was present in urea at a concentration of 2.5%. This effect is well known to manufacturers who closely monitor the presence of the compound in urea formulations. Today, concentrations are usually well below 1% biuret in urea and the problem is largely of academic interest. It should be noted, however, that specifications for urea for foliar application call for 0.5% of less biuret.

Nitrogen Solutions

Nitrogen solutions have dramatically increased in popularity in the last 20 years. Two major classes of nitrogen solutions exist. This classification is based upon the presence of free ammonia or a positive vapor pressure of ammonia. Solutions which contain free ammonia are classified as 'pressure solutions' while those without free ammonia are 'non-pressure'. Non-pressure solutions are utilized in largest quantities world-wide. They are comprised primarily of ammonium nitrate and urea but can contain other compounds such as ammonium sulfate and calcium nitrate. Pressure solutions must be applied to the soil or in irrigation water in a manner similar to anhydrous ammonia due to the presence of ammonia. Pressure solution components include ammonia, ammonium nitrate, urea, and possibly ammonium sulfate or calcium nitrate.

Theoretically, higher nitrogen concentrations are associated with pressure solutions. However, the greatest usage in the United States of pressure solutions is as aqua ammonia which contains only 20% nitrogen. On the other hand, non-pressure solutions in most common use contain from 28 to 32% nitrogen. The higher nitrogen concentration of these solutions coupled with their greater adaptability to different types of application including application in irrigation water, combinations with herbicides, and direct surface applications dictates their greater popularity.

The production of nitrogen solutions involves no special chemistry since the three major components have already been discussed. A peculiar chemical quirk does exist in the production of these solutions, however, particularly non-pressure solutions. Solutions of either ammonium nitrate alone or urea alone fail to achieve the nitrogen concentration that results from a mixture of the two materials (see Table 12). Solubility of both urea and ammonium nitrate is enhanced by the presence of the other compound. The solubility of ammonium nitrate is 118.3 g/100 ml of water at 0C. The solubility of urea, on the other

Table 12. Composition and nomenclature of pressure and non-pressure nitrogen solutions.

Solution nomenclature**	Classification	Composition % by weight			Salt-out temp., C*
		$\%NH_3$	$\%NH_4NO_3$	% Urea	
160(0-46-0)	Non-pressure	0	46	0	-12
200(0-57-0)	Non-pressure	0	57	0	5
200(0-0-44)	Non-pressure	0	0	44	5
230(0-0-50)	Non-pressure	0	0	50	17
245(0-70-0)	Non-pressure	0	70	0	30
280(0-39-31)	Non-pressure	0	39	31	-18
320(0-44-35)	Non-pressure	0	44	35	0
201(24-0-0)	Pressure	24	0	0	-52
410(19-58-11)	Pressure	19	58	11	-14
410(26-56-0)	Pressure	26	56	0	-31

* Salt-out temperature is that temperature where crystal formation occurs in solution. Note that salt-out temperature increases as the nitrogen concentration increases (non-pressure solutions).

** The first number outside parenthesis refers to the % total nitrogen (without decimal; 160 = 16.0% nitrogen). Numbers inside parentheses refer to components of the solution.

hand, is 78 g/100 ml of water at 5C. In the presence of the other compound the solubilities of ammonium nitrate and urea have been changed to 130 g/100 ml and 103 g/100 ml at 0C, respectively.

Use of Nitrogen Solutions. Use of nitrogen solutions is determined entirely by the type of solution being considered. As noted earlier, non-pressure solutions are much more flexible in terms of application techniques than are pressure solutions. Urea-ammonium nitrate solutions (UAN) are adapted to a number of application techniques. In irrigated agriculture, water application of UAN solutions (fertigation) has become increasingly popular as a means of improving nitrogen use efficiency, and elimination of one or more trips over the field. Fertigation has developed relatively recently although as noted in the section on use of ammonia, the process dates back to the 1930's.

The University of Nebraska has the leader in the development and study of fertigation. Paul Fischbach and colleagues in Agricultural Engineering at Nebraska began fertigation research in 1960. Their work has shown that nitrogen use efficiency can be improved by applications through the water. Losses due to leaching are minimized by split or multiple nitrogen applications as compared to applications prior to seeding particularly on sandy soils. Data in Table 13 indicate the effect that fertigation can have on corn yields (Fischbach, 1964).

Table 13. Corn yields for application of nitrogen by mechanical means and for application in the irrigation water (Fischbach, 1964).

Method of application	Yield kg/ha
Normal mechanical application[a]	7275
Normal (mechanical and irrigation split)	7526
Normal irrigation (all in one irrigation)	7777
Normal irrigation (split into two irrigations)	7840
Normal mechanical plus 20 kg of nitrogen in irrigation water	8216

[a] Normal-recommended amount of nitrogen determined by a soil test and past cropping history.

Widest adaptation for this technique has occurred in sprinkler irrigated sandy soils. Estimates indicate that about 60-70% of the sprinkler irrigation systems in the Colorado, Kansas, Nebraska, Oklahoma area involve application of nitrogen and/or other nutrients through the irrigation water. This technique is also adaptable to gated pipe or ditch irrigation but development there has not been quite so widespread as with sprinkler systems.

Fischbach (1970) examined the distribution pattern of nitrogen solutions applied through sprinkler and gated pipe irrigation systems. Since nitrogen applications could be affected by improper or incomplete mixing in the water, tests were conducted for uniformity of nitrogen concentrations at various points on a gated pipe system and at points along a center pivot sprinkler system. Data reported in Tables 14 and 15 show good uniformity indicating good mixing.

Timing nitrogen solution application in water under furrow irrigated conditions is important in maintaining a good distribution. Fischbach (1964) suggested that acceptable nitrogen distribution could be achieved by using as large an irrigation stream in each furrow as possible without causing serious erosion on the particular slope being irrigated (Table 16). He suggested that the stream should reach the far end of the furrow in a maximum recommended time for various soil textures (Table 17). If it does not reach the end of the furrow in the time indicated, the run may be too long, water penetration is uneven and nitrogen distribution suffers. Fischbach notes that if the water flows through the field in less than the recommended maximum time, the distribution will be still better. For efficient use of fertilizer nitrogen, Nebraska researchers suggest that the nitrogen solution should be injected into the water as soon as the stream is started down the furrow and continue until the water and fertilizer reach the far end of the field in all furrows. The nitrogen should then be shut off and irrigation continued until water penetrates to the desired depth.

Table 14. Concentration of nitrate-nitrogen in the irrigation water from the 4th, 11th, 23rd and 29th gate openings at various intervals of time in a gated pipe irrigation system (Fischbach, 1970).*

Elapsed Time Minutes	Gate Number			
	5th	11th	23rd	29th
	------------Nitrate-nitrogen, mg/l-----------			
10	27.2	27.2	27.2	27.2
20	27.2	28.8	27.2	27.2
30	28.8	28.8	28.2	27.2
40	27.2	28.8	28.2	27.2

* A 4000 liter per minute system, using 20 cm gated pipe 384 m. long. The water pressure at the beginning of the gated pipe was 528 g/cm^2.

Table 15. Concentration of nitrate-nitrogen in irrigation water caught half-way between towers of a 12 tower center-pivot sprinkling system (Fischbach, 1970).*

Location of Sample	Nitrate-nitrogen mg/l
Pivot Point and Tower No. 1	35
Tower No. 1 and 2	35
Tower No. 2 and 3	35
Tower No. 3 and 4	32
Tower No. 4 and 5	36
Tower No. 5 and 6	34
Tower No. 6 and 7	33
Tower No. 7 and 8	34
Tower No. 8 and 9	35
Tower No. 9 and 10	34
Tower No. 10 and 11	37
Tower No. 11 and 12	34

* A 3600 l/minute system, 392 m. long using 17-cm diameter OD pipe, 4.22 kg/cm^2 pressure at the pivot. Fertilizer solution (UAN) injected by positive displacement pump.

Table 16. Maximum furrow stream size for various slopes (Fischbach, 1970).

Percent slope	Liters per minute per furrow
1.0	40
0.5	80
0.3	120
less than .3	200

Table 17. Approximate maximum length of time for water to flow to the end of the field on various soil textures (Fischbach, 1970).

Soil texture	Hours
Loamy sands	2-3
Sandy loams	3-4
Fine sandy loams	4-5
Silt loams	5-6
Silty clay loams	6-7

Use of nitrogen solutions or other nitrogen sources in irrigation water should include a re-use system for tailwater in order to protect against contamination of surface water with nitrogen and to make efficient use of both nitrogen and water.

Non-pressure nitrogen solutions are most commonly used for fertigation because they avoid problems of calcium salt precipitation produced by ammonia and avoid problems of ammonia loss through sprinklers (see ammonia use section). Pressure solutions would be most adapted to ditch-siphon tube irrigation systems.

Nitrogen solutions can be used in dual application with herbicides. This process results in elimination of a trip over the field with savings in time and fuel. Herbicides applied in this manner have been effective as long as the placement and timing were compatible with recommendations for the herbicide. Some suggestions have been made that the activity of the herbicide is actually improved due to the more complete coverage generated by the higher amount of solution applied.

Comparisons of the agronomic effectiveness of nitrogen solutions versus ammonia or the various solid materials have indicated good performance. Nitrogen solution management, like that of urea, can possibly be improved at times by incorporation, particularly when applied to high pH soils where ammonia volatilization from urea could be occurring at the soil surface. When irrigation water is applied shortly after application, incorporation may be less important. Incorporation of nitrogen and other elements is always a good idea if adaptable to the operations.

In summary, recommendations from most sources do not distinguish between nitrogen fertilizers but leave that decision to the producer based on applied cost and how the particular type of nitrogen fits into the operation. Iowa data presented in Table 18 (Voss, 1978) are an excellent indication of the uniformity which exists between performance of nitrogen sources.

Table 18. The influence of source, time, and rate of nitrogen fertilizer application on corn yields (Voss, 1978).*

Nitrogen Treatment			Corn Yields, kg/ha	
Rate kg/ha	Time	Source	Nashua	Kanawha
0			4077	4578
90	Early	Am. nitrate	6397	7903
		Solution 28	6209	7589
		Urea	6334	8028
90	Late	Am. nitrate	6586	7840
		Solution 28	6021	7714
		Urea	6648	8091
180	Early	Am. nitrate	6711	8655
		Solution 28	6586	8718
		Urea	6397	8404
180	Late	Am. nitrate	6397	8593
		Solution 28	6586	8593
		Urea	5958	8781

* From data courtesy of Dr. John Webb, Iowa State University.

Ammonium Sulfate

Ammonium sulfate ($(NH_4)_2SO_4$) has a long history as a nitrogen source. This material has been manufactured as a by-product from several industrial processes. At one time, ammonium sulfate represented the most common source of nitrogen in many sections of the world and still remains an important source particularly in rice producing areas. A major disadvantage to its use is its relatively low nitrogen content (21%) and the acid residue it leaves in the soil.

Ammonium sulfate is a common by-product of the steel industry particularly the coking of coal when some ammonia is present in the coke-oven gas. Sulfuric acid, another common by-product material, can be marketed by conversion into ammonium sulfate either by reacting with anhydrous ammonia, aqua ammonia or waste gas streams which may contain some free ammonia. Waste ammonia is frequently recovered in this manner from the production of synthetic fibers. Production chemistry is very simple:

$$2NH_3 + H_2SO_4 \longrightarrow (NH_4)_2SO_4.$$

Whatever the source, ammonium sulfate's physical and chemical characteristics remain the same. These characteristics are set out in Table 19.

Table 19. Properties of ammonium sulfate.

Color	White crystalline; varies with presence of contaminants
Molecular weight	132.14
Density	1.769
Melting point	Decomposes at 235C
Solubility	70.6 g/100 ml at 0C 103.8 g/100 ml at 100C
Nitrogen, % (pure)	21.2
Sulfur, % (pure)	24.2
Nitrogen, % (fertilizer grade)	21
Sulfur, % (fertilizer grade)	24

The color of solid ammonium sulfate varies depending upon its source. Ammonium sulfate produced from petroleum by-products frequently has a dark color due to the presence of hydrocarbons or coal tars. United States consumption of ammonium sulfate was 1.82 million metric tons in 1977.

Use of Ammonium Sulfate. Ammonium sulfate has proven to be an excellent nitrogen source. Obviously, an advantage for ammonium sulfate might exist in its sulfur content if that element were in short supply. Ammonium sulfate contains about 24% S.

Residual effects of ammonium sulfate on the soil were noted earlier as being more acidic than residual effects of other nitrogen sources. Guidelines for the soil acidification produced by ammoniacal nitrogen materials are outlined in Table 20.

Table 20. Equivalent acidity or alkalinity of nitrogen fertilizers expressed in kg pure calcium carbonate required to neutralize residual acidity.[1]

Materials	Nitrogen %	Per kg of ammonium nitrogen	Per 20 kg nitrogen	Per 100 kg Material
Ammonium sulfate	21	5.35	107	110
Monoammonium phosphate	11	5.00	100	55
Anhydrous ammonia	82	1.80	36	148
UAN Solution	28	1.80	27	38
39% ammonium nitrate				
31% urea				
Presure Solution	41	1.80	27	75
19% NH_3				
58% ammonium nitrate				
11% urea				
Ammonium Nitrate	34	1.80	18	15
Urea	46	1.80	36	84

Calcium carbonate equivalent added to the soil

	Nitrogen %	Per kg nitrogen	Per 20 kg nitrogen	Per 100 kg Material
Calcium nitrate	15.5	1.35	27	21
Sodium nitrate	16.0	1.80	36	29

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[1] Adapted from Tisdale and Nelson, 1975.

According to the theory of Pierre, application of a kg or nitrogen as ammonium sulfate requires approximately 5 kg of effective calcium carbonate to offset the residual acidity from the nitrification of the ammonium nitrogen and the sulfuric acid formed. Compare these values with those for urea, anhydrous ammonia and ammonium nitrate.

Continued use of ammonium sulfate for extended periods of time without adequate liming can result in drastically reduced plant growth. The high acidity produced results in the destruction of clay minerals and subsequently the release of large amounts of aluminum which is toxic to plants. In addition, phosphorus fixation may be enhanced due to the presence of large amounts of soluble iron and aluminum in the soil.

Ammonium sulfate has fallen to a rather insignificant position in the total supply of nitrogen in the U.S. That is not to say that it is not used but is frequently used in the manufacture of mixed fertilizers, both solids and liquids. Its primary disadvantage aside from the acidity mentioned above is its low nitrogen content, only about 20-21%, depending upon the source. Freight rates then tend to increase the cost per unit of nitrogen.

Agronomic comparisons of ammonium sulfate and other nitrogen sources have usually indicated agronomic equality. Some volatilization losses of ammonia can occur under certain conditions. This is not necessarily a characteristic of ammonium sulfate since factors other than nitrogen source control the process.

Sodium Nitrate and Calcium Nitrate

Sodium nitrate ($NaNO_3$, 16.5% nitrogen) was the first commercial source of nitrogen utilized in the United States. This material, imported from Chile, was the nitrogen source for the cotton industry in southeastern United States.

Sodium nitrate is produced by processing a natural ore mined from high, dry Andean valleys. The source of the nitrate in these deposits is of debatable origin but one of the prominent theories relates to its derivation from fumaroles associated with volcanic activity. The nitrogen was eventually nitrified and deposited beneath detritus washed down from surrounding slopes. The dry climate of the region protected the highly soluble nitrate salts. This deposit is unique in the world in terms of its extent and accessibility.

Today, sodium nitrate is produced from the Chilean deposits as well as from other industrial sources. Major disadvantages include low nitrogen content, high freight costs associated with movement of the natural material from Chile, and the sodium content. It assumes a very minor role in terms of nitrogen materials in the United States today.

Calcium nitrate ($Ca(NO_3)_2$, 15.5% nitrogen) was one of the first commercial fertilizers manufactured in the 20th century from synthetically fixed nitrogen. The process developed in Norway was dependent upon the production of nitric acid from electric arc fixation of elemental nitrogen and is essentially identical to the mechanism described for nitrogen fixation by lightning.

Today, calcium nitrate is produced in Norway from nitric acid produced from anhydrous ammonia. The disadvantage of the use of calcium nitrate is the high cost per pound of nitrogen which is largely due to freight required to move the material from Scandinavia to the United States. Agronomically, this material is an excellent source of nitrogen and in addition to providing soluble nitrate also provides a readily available source of calcium to plants. A characteristic of both sodium and calcium nitrate is an alkaline residual effect on the soil. Continued use will tend to raise soil pH.

Slow Release Nitrogen Fertilizers

Low percentage recovery of applied nitrogen has been recognized for years. Results of field and lysimeter studies have shown that 70% or less of applied nitrogen is recovered by crops. Reasons for poor recovery of highly soluble nitrogen sources include ammonia volatilization, leaching of nitrate and denitrification. Large amounts of available nitrogen in the soil at one period in time may also lead to luxury nitrogen consumption by plants which represents absorption above that necessary for normal plant growth. Slowing the rate of release of nitrogen from fertilizers is a conceivable means of avoiding these problems.

The problem of slowing nitrogen release from fertilizers has been approached by altering physical characteristics of fertilizers and by developing compounds which have low water solubility. The release of nitrogen from soluble materials has been altered by coating water soluble compounds with less water soluble materials which retard entry of water into the particle and movement of nitrogen out. Coatings applied to soluble nitrogen materials generally have been of three types:

1. Coatings with small pores which allow slow entrance of water and slow passage of solubilized nitrogen compounds out of the encapsulated area;
2. An impermeable coating that requires breakage by physical or chemical action before the nutrient is dissolved; and
3. Semi-permeable coatings through which water diffuses and creates internal pressure sufficient to disrupt the coating.

Sulfur-coated urea (SCU) has been developed in recent years as a product with the characteristics of slow nitrogen release. This material is basically urea with a coating of elemental sulfur including a binding agent, a sealant and a microbiocide. Elemental sulfur was chosen as a coating agent because of its relatively low cost and ease of handling. Nitrogen release rates are varied by controlling the thickness of the coating. The nitrogen content of SCU ranges from about 30 to 37% dependent upon the thickness of the sulfur coating. Nitrogen release is calculated as the amount of the nitrogen that will solubilize in seven days. Characteristics of SCU which have been examined by the Tennessee Valley Authority include material ranging from 10% nitrogen solubilized in the first seven days up to 40% nitrogen solubilized in the first seven days. These forms of SCU are noted as SCU-10 and SCU-40.

Agronomic evaluations of SCU have indicated some possible improvement in nitrogen efficiency particularly with crops with very large demands for nitrogen throughout an extended growing season. Rice, bermudagrass and sugarcane have been crops where positive responses to SCU versus conventional urea have occurred. (Uchida et al, 1975; Shelton, 1976; Snyder and Gascho, 1976; Gascho and Snyder, 1976; Diamond, 1975; Dalal, 1974 and 1975).

Uncoated organic compounds of low water solubility have been studied as sources of nutrients for over 50 years. Intensive studies during the last 25 to 30 years have resulted in the production of certain urea-derived compounds which have a characteristic of slow nitrogen solubility in water. Typical of these types of compounds are those produced by the reaction of urea and formaldehyde to produce condensation products known generally as urea-formaldehyde or urea-form.

Urea reacts with the formaldehyde in the presence of a catalyst to form a mixture of compounds. These materials are white, odorless compounds which contain varying amounts of nitrogen but averaging around 38%. A basic component of urea-form is methylene-urea polymer varying in chain length and in degree of cross-linking between chains. Basically, as the compound links are extended, the water solubility of these materials declines.

Although the urea-forms have been examined agronomically in the United States, they have not received wide attention except as nitrogen sources for very high value crops, turf and ornamentals. More attention has been given to these compounds in Europe and in Japan.

Isobutylidenediurea (IBDU) is a reaction product of urea and isobutyraldehyde. The reaction has been utilized in Germany and Japan to produce a compound containing approximately 32% nitrogen and having a low water solubility (0.1 -0.01%). This compound also has the ability to be granulated with soluble nutrient sources of phosphorus and potassium to supply mixed fertilizer formulations with slow nitrogen release.

Crotonylidene diurea (CDU) is another urea based material which is produced in Germany, Japan and France from crotonaldehyde and urea. The nitrogen content of this material is about 32%.

Other slow release nitrogen compounds have been studied or produced in small quantities in some countries. Basically, the characteristics of these materials involve high cost per unit of nitrogen. Use should be considered as being relegated to very high value crops, ornamentals and turf. Extensive agronomic use has not been demonstrated to date as being economically feasible in the U.S.

ORGANIC NITROGEN FERTILIZERS

In addition to the manufactured nitrogen sources, organic, nitrogen-containing compounds available for fertilizer use include bird guano, poultry and animal manures of all types and municipal sewage sludges (Table 21). The analyses of these materials vary widely depending upon their source and the amount of weathering which has occurred prior to their being collected for fertilizer use. Bird guano has one of the longest histories of direct use and contains not only nitrogen but appreciable quantities of phosphorus and other plant nutrients. Supplies of guano are relatively scarce and are not used extensively today.

Animal wastes as sources of nutrients for crops have long been recommended because of the efficiency involved in re-cycling nutrients and the fact that the materials being added to the soils contain not only nitrogen but carbon and many other plant nutrients. Animal wastes vary widely in composition depending upon the composition of the rations, types of animals involved and the storage conditions for manure prior to its application. Some indication of the range in nitrogen concentrations of various animal wastes can be obtained from Table 22.

Studies of the availability of nitrogen from animal waste applica-

Table 21. Average composition of some common organic materials.

Source	Percent		
	N	P	K
Activated sewage sludge	6.0	0.9	–
Blood, dried	13.0	–	–
Bone meal (raw)	3.5	9.5	–
Bone meal (steamed)	2.0	12.2	–
Castor pomace	6.0	0.6	0.4
Cocoa meal	4.0	0.6	2.1
Cocoa shell meal	2.5	0.4	2.5
Cocoa tankage	2.5	0.6	1.0
Cottonseed meal	6.6	1.1	1.2
Fish scrap (acidulated)	5.7	1.3	–
Fish scrap (dried)	9.5	2.6	–
Garbage tankage	2.5	0.6	0.8
Peanut meal	7.2	0.6	1.0
Peanut hull meal	1.2	0.2	0.7
Peat	2.7	–	–
Peruvian guano	13.0	5.4	2.1
Process tankage	8.2	–	–
Soybean meal	7.0	0.5	1.2
Tankage, animal	7.0	4.3	–
Tobacco stems	1.5	0.2	4.2
Whale guano	8.5	2.6	–

Tisdale and Nelson (1975).

Table 22. Composition of various manures and waste materials.

	Dry Matter %	N %	K %	P ppm
Beef feedlot manure[1]				
High values	38.5	1.08	0.95	3,680
Low values	24.2	0.44	0.58	2,100
Mean values	32.4	0.71	0.74	2,830
Liquid poultry manure[2]	8.1	0.16	0.24	180
Liquid swine manure[1]	0.6	0.01	0.06	20
Liquid swine manure[1]	2.6	0.09	0.07	239
Liquid dairy manure[1]	8.6	0.24	0.19	210
Dairy cattle manure[2]	21	0.56	0.50	1,000
Sheep manure[2]	35	1.40	1.00	2,100
Swine manure[2]	25	0.50	0.38	1,400
Poultry manure[2] (w/o litter)	46	1.56	0.35	4,000
Poultry manure[3]				
High values	26	1.50	0.74	7,100
Low values	28	1.00	0.70	6,800
Activated sewage sludge[4]	100	5.56	0.36	25,860
Lagoon sewage sludge[4]	100	1.71	0.23	18,750

[1] Murphy, L.S., W. L. Powers and H. L. Manges, unpublished data, Kansas State University, 1971.

[2] Benne, E.J., C. R. Hoglund, E. D. Longnecker, R. L. Cook, Michigan Agricultural Experiment Station Circular Bulletin 231, 1961.

[3] Robertson, L.S., Department of Crop and Soil Sciences, Michigan State University. Manure utilized in fertilization study with corn, 1 source.

[4] Knezek, B.D., Department of Crop and Soil Sciences, Michigan State University.

tions have indicated that not all the nitrogen obtained in the wastes is available the first year. From nitrogen contents of around 1% in feedlot wastes, calculations from California, Kansas and Wisconsin, have indicated that about 13 kg of nitrogen will be available the first year per metric ton of applied animal waste on a dry matter basis. This will be a function of the climate, the original composition of the manure and storage. Data from Texas have indicated generally higher concentrations of nitrogen in beef feedlot wastes if the material is collected and applied after short storage periods. Iowa research and extension personnel have calculated nitrogen availability from several types of animal wastes (Table 23) (Voss, 1978).

Table 23. Nutrient content per metric ton of solid manure from solid handling systems (Voss, 1978).

Animal	Bedding or litter	Dry matter	Available nitrogen	Total nitrogen	Phosphorus	Potassium
		%	----------kg/metric ton of manure---------			
Swine	No	18	3	5	2	3.3
	Yes	18	2.5	4	1.5	3
Beef cattle	No	15[a]	1.5	5.5	1.5	4
	Yes	52[b]	3.5	10.5	3	9.5
		50	4	10.5	4	10.8
Dairy cattle	No	18	1.5	4.5	1	4
	Yes	21	2.5	4.5	1	4
Sheep	No	28	1	9	2.4	10.8
	Yes	28	1	7	2	10.4
Poultry	No	45	13	16.5	10.4	14
	Yes	75	18	28	9.8	14
	Deep pit	76	22	34	14	18.7
Turkey	No	22	8.5	13.5	4.3	7
	Yes	29	6.5	10	3.5	5.4
Horse	Yes	46	2	7	1	5.8

Source: Livestock Facilities Handbook, 1975. Iowa State Univ. Coop. Ext. Serv. MWPS-18. a. Open concrete lot. b. Open dirt lot.

Applications of animal wastes as sources of nitrogen for irrigated crops should be dictated by the amounts of nutrients needed and the amount calculated to become available from a waste source during the course of the growing season. Guidelines for such applications have been formulated in several states (Powers et al 1973, 1974). Excessive rates of application of wastes (solids and liquids) can lead to many problems in crop production including plant damage from accumulated salts and high accumulations of nitrate nitrogen in forages and in the soil. Adriano et al (1971) studied this problem in California. Wallingford et al (1974 and 1975) noted high accumulations of potassium and ammonium cations in soils receiving large amounts of animal wastes. Severe yield depressions resulted from the salt accumulations. Had recommended rates of application been followed, however, these problems would not have occurred. Waste disposal and waste utilization management are vastly different concepts. Soil incorporation of wastes shortly after application will improve nitrogen use efficiency by avoiding volatilization losses of ammonia.

Municipal wastes also contain nitrogen and have been utilized extensively all over the world as nitrogen supplies for crops. Highly processed and leached sewage sludge (solids) may contain low amounts of nitrogen, frequently less than 1%. Nitrogen has been lost by volatilization of ammonia or has been leached out into the liquid portion of the waste. This is a fate of much of the nitrogen in animal and human wastes since nitrogen is largely voided in the form of urea.

Seaweed has been utilized as a source of nitrogen under some conditions where it has been harvested for extraction of certain chemicals. Composition will vary depending upon the source and this practice is less widespread than the use of animal manures for nitrogen. Effective use of any waste material as a fertilizer demands some familiarity with the composition of the material. Analyses are urged where possible and when time permits.

DETERMINING NITROGEN NEEDS

Soil analysis has played an increasingly important role in determination of nitrogen needs of irrigated crops in the past few years. In most states in the western half of the U.S., state and commercial soil testing services are now offering a residual test for nitrate-nitrogen. Accumulations of nitrate in the soil from whatever source can be effectively utilized by the following crop, lowering the amount of additional nitrogen needed for the particular yield goal and diminishing the amount of nitrogen that may leach through the soil profile and into shallow aquifers. Nitrogen normally does not store well in the soil and should be utilized by following crops. Soil analysis is the only means of determining the magnitude of such accumulations. Analyses for total soil nitrogen do not provide an accurate measurement of the relatively small quantities of available nitrogen which have accumulated.

Continued adaptation to good management practices already recommended should allow maximum production from irrigated areas without environmental degradation. Agricultural research will continue to be the mainstay of both production of crops and environmental quality.

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MINERALIZATION, IMMOBILIZATION AND NITRIFICATION

F. E. Broadbent[1]

ABSTRACT

The decomposition of organic substances containing nitrogen through the activities of soil microorganisms and resulting in the release of some of the nitrogen as ammonia is called nitrogen mineralization. Net mineralization occurs when the quantity of nitrogen in the material undergoing decomposition exceeds the needs of the microbial population for nitrogen to produce new cells. If the decomposing substances do not contain enough nitrogen to meet the needs of the microbial population, any inorganic nitrogen present in the soil will be utilized by the microbes and converted to cell protein and other nitrogenous compounds. This process is called immobilization. In a sense the assimilation of inorganic nitrogen by growing plants is also immobilization, but this discussion considers only the microbiological process.

The nitrogen supplying capacity of soils depends to a large extent on rates of mineralization and immobilization. Mineralization must exceed immobilization in order for nitrogen to be made available for crop plants. Various procedures have been utilized to obtain estimates of nitrogen mineralization over a growing season, the most successful of which have been based on incubation of soil samples under controlled conditions for a few weeks.

The oxidation of ammonium nitrogen by certain species of soil bacteria is called nitrification. Nitrification is seldom a limiting

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step with regard to the capacity of a soil to produce available nitrogen, but it has an important influence on fertilizer efficiency and movement when ammonic fertilizers are applied. Nitrification rates are affected somewhat by environmental parameters such as temperature and pH.

Desirable management practices should favor not only efficient utilization of fertilizer nitrogen, but also maximum crop uptake of mineralized nitrogen. Experiments with isotopically labeled fertilizers indicate that these two objectives are compatible with each other and also with the need to minimize leachable nitrate.

INTRODUCTION

The process of mineralization, involving the gradual decomposition of organic forms of nitrogen with the release of ammonia, largely determines the nitrogen fertility status of a soil. The reverse process of immobilization, whereby inorganic forms of nitrogen such as ammonium and nitrate are converted into organic forms, may predominate for a short period of time, but the normal situation is that mineralization is the more rapid of the two. This makes possible a continued slow release of mineral or available nitrogen which permits plants to grow. Plant uptake of mineral nitrogen is a form of immobilization, but will not be discussed in this paper.

Once ammonia is produced, it is readily converted to nitrate by specialized bacteria which are abundant in most soils. This process is called nitrification. The term nitrification is sometimes applied to the entire sequence of events starting with organic nitrogen and ending with nitrate, but correctly used, it applies only to the oxidation of ammonia to nitrate. Somewhat illogically, the reverse of nitrification is not called denitrification, but nitrate reduction, since the term denitrification through common usage has come to mean a special type of nitrate reduction resulting in evolution of gases. The subject of denitrification is discussed in detail in another paper in this volume.

We can depict the transformations under consideration by means of a simple diagram as follows:

$$\text{Soil Organic N} \underset{\text{immobilization}}{\overset{\text{mineralization}}{\rightleftharpoons}} NH_4^+ \underset{\text{nitrate reduction}}{\overset{\text{nitrification}}{\rightleftharpoons}} NO_3^-$$

In the soil system the forward and reverse reactions all proceed simultaneously, but at different rates. The reaction rates may be altered considerably by environmental conditions and soil management. If we let m = mineralization rate, i = immobilization rate, n = nitrification rate, and r = nitrate reduction rate, the usual situation is that $m > i$, $n > m$, and $n \gg r$. The practical consequence of these relationships in agricultural soils is a continued conversion of organic nitrogen to nitrate, with little or no accumulation of ammonium. Although this is the usual situation, it is not the invariable one, or all organic nitrogen would eventually be converted to nitrate. This paper will discuss how these interrelated reactions are affected by soil conditions and management practices.

MINERALIZATION-IMMOBILIZATION

It is necessary to consider mineralization and immobilization together, since one does not occur without the other. Mineralization is the result of microbial decomposition of nitrogen-containing organic materials such as plant residues and soil humus, which are used by the microbes as a source of energy. At the same time, the development of microbial cells requires carbon and nitrogen, along with other essential elements, which most microorganisms obtain from the substances undergoing decomposition. If there is more nitrogen present in these substances, called the substrate of the microorganisms, than meets their needs, some is released as a waste product in the form of ammonia, or more correctly, of ammonium, since in the presence of water ammonia forms the ammonium ion. In this situation net mineralization occurs, as is illustrated in Figure 1 (a). On the other hand, if there is insufficient nitrogen present to satisfy the needs of the soil organisms, as is often the case when a highly carbonaceous material is undergoing rapid decomposition, any inorganic nitrogen present will be assimilated by the organisms as rapidly as it is produced and net immobilization will occur. Usually the rate of immobilization will not exceed the rate of mineralization for long periods of time. Numbers of microorganisms will increase very rapidly while a source of energy is available, but when the readily available fraction of the energy source becomes depleted, numbers of microorganisms decline rapidly to a much lower level. When this occurs some of the nitrogen previously locked up in microbial cells becomes surplus as the cells die off and are themselves decomposed. This nitrogen is released as ammonia. The rapid turnabout from a period of net immobilization where $m<i$ to one of net mineralization where $m>i$ is illustrated in Figure 1 (b). This pattern is typical of that which occurs when a mature crop residue is turned into the soil. During the initial period of net immobilization inorganic nitrogen supplied as fertilizer would be converted to organic nitrogen in the form of microbial protein, thereby making it inaccessible to crop plants. However, after the elapse of a few weeks the supply of inorganic nitrogen may attain a value as high as or higher than before the addition of the organic residue.

Influence of Environmental Factors on Immobilization and Mineralization

It is important to recognize that some interchange of nitrogen between the organic and inorganic forms occurs whenever environmental

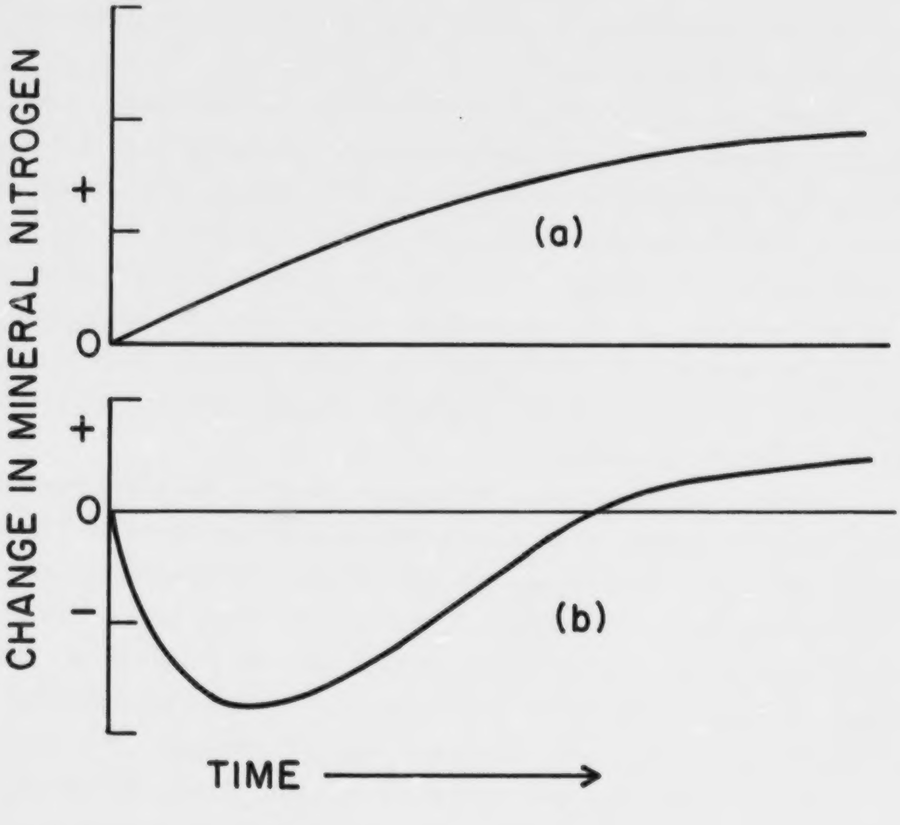

Figure 1. Changes in mineral nitrogen in a situation of continuous net mineralization (a), and a period of net immobilization followed by net mineralization, (b).

conditions permit microbial activity. Although in general microbial activity is favored by those conditions which are optimal for crop growth, microorganisms will usually tolerate a wider range of environmental conditions than will higher plants. The relative rates of immobilization and mineralization depend primarily on the kind and amount of organic energy material or substrate present in the soil, but are also significantly affected by such factors as temperature, moisture, aeration and pH.

Temperature. When organic residues of low nitrogen content undergo decomposition, net immobilization occurs, but the total amount of nitrogen eventually immobilized is not strongly temperature dependent. The lower the temperature, the longer the period of net immobilization (Broadbent, 1968). Under conditions such that net mineralization is favored, the rate increases with increasing temperature up to a maximum at about 40C. The poor nitrogen-supplying power of cold soils is a reflection of the influence of low temperature in retarding rates of mineralization.

Moisture and Aeration. Within the range of moisture contents which provide enough water for microbial activity there is little influence of moisture on quantity of nitrogen immobilized. The effect of excess moisture sufficient to restrict aeration is to induce a qualitative change in the population of soil microorganisms. Whereas adequate oxygen permits the development of a heterogeneous population of bacteria, fungi, algae, actinomycetes and protozoa, the exclusion of oxygen inhibits the development of aerobic forms, particularly the fungi and actinomycetes. These two groups make up a large part of the total mass of living cells in soil, or biomass, and their inhibition results in a substantial reduction in the amount of nitrogen needed to support the soil population. Another consideration is that aerobic metabolism in microorganisms is inherently more efficient in cell synthesis than is anaerobic metabolism. Thus, as the soil moisture content approaches saturation there is a decrease in the immobilization rate, while mineralization rate is less affected. For this reason net mineralization may be greater in wet or flooded soils than in those same soils at field capacity. This is not to say that saturated soils have less need for fertilizer nitrogen, as for example in rice culture, because the mineralized nitrogen is also more likely to be lost through denitrification or other mechanisms.

Nitrogen mineralization can proceed slowly in soils too dry for crop growth. Stanford and Epstein (1974) reported that in soils ranging from sandy to silty clay loams, mineralization of nitrogen was linearly related to per cent soil water in the range of moisture contents between field capacity and the wilting percentage.

Birch (1960) reported that when soils are re-wetted after a period of drying there is an increase in the rate of nitrogen mineralization in comparison with soils which have been maintained in the moist condition. The magnitude of the increase is related to the organic matter content of the soil and to the length of the drying period. He reported increases in mineralization of nitrogen ranging from 32 kg/ha in soils containing 1% carbon and which had been air-dried for three weeks, up to 133 kg/ha in soils containing 5.5% carbon. This phenomenon is probably not of great importance during growth of crops because the degree of drying required would be attained in a shallow surface layer at most. However, it may account for initial flush of available nitrogen when rains or irrigation wet the soil after a fallow period in dry weather.

pH. The optimum pH for most soil organisms is in the neutral to slightly alkaline range. Immobilization and mineralization rates are therefore highest in this range. However, immobilization rates are affected by the nature of the inorganic nitrogen present and its relationship to soil pH. Unlike most crop plants, soil microorganisms utilize ammonium nitrogen preferentially over nitrate, but the quantity of either ammonium or nitrate immobilized in a given situation is affected by soil pH (Broadbent and Tyler, 1965). As the pH increases above 7, the amount of ammonium nitrogen immobilized increases because the effect of ammonium assimilation is to lower the pH. Conversely, as the soil becomes more acidic there is a tendency for immobilization of nitrate to increase because the effect of its utilization is to increase pH.

Quantities of Nitrogen Immobilized

Immobilization of nitrogen is commonly discussed in terms of carbon/nitrogen ratios, which gives a quantitative expression of the relationship between the size of the food supply represented by a given organic substance and the nitrogen present in it. Since the carbon content of most organic residues of the kind that find their way into soils is usually in the range of 40 to 45% there is little to be gained by using carbon/nitrogen ratios as compared with a simple expression of nitrogen content. A fairly good rule of thumb is that mature plant materials containing less than 1.3-1.5% nitrogen on the dry weight basis can be expected to immobilize additional nitrogen during the initial stages of decomposition. Residues such as straw and stubble of small grains will immobilize up to 10 kg nitrogen per metric ton (mt) of residue, corn stalks about 8 kg/mt, and sawdust and rice hulls about 5 kg/mt. The latter have very low nitrogen contents, but immobilize less nitrogen than would be expected because of their slow rate of decomposition.

In the decomposition of carbonaceous organic materials where supplemental inorganic nitrogen is not available, the decomposition rate is temporarily retarded by lack of nitrogen because the soil population does not attain the size it would have done had sufficient nitrogen been present. Even in this condition the period of net immobilization seldom lasts more than a few weeks, and the reestablishment of net mineralization is simply delayed.

A factor which complicates attempts to establish a quantitative relationship between residue composition and the amount of nitrogen actually immobilized is the utilization of excess nitrogen by microorganisms when a large supply of inorganic nitrogen is available. When varying levels of nitrate were added to a soil along with corn stover containing 0.83% nitrogen, Munson and Pesek (1958) found that net immobilization was directly related to the amount of nitrogen added, but the additional nitrogen taken up through luxury consumption was rapidly re-mineralized, as illustrated in Figure 2.

Release of Immobilized Nitrogen

Since immobilization of inorganic nitrogen represents a diversion of this plant nutrient from its intended use, it is a matter of practical importance to know something about the length of time which elapses before immobilized nitrogen is again converted to the mineral form. This information can be obtained conveniently when a labeled source of inorganic nitrogen is present in the soil during a period of net immobilization. The re-conversion of labeled organic nitrogen to the inorganic form appears to be a very slow process, sometimes requiring years, or even decades (Broadbent and Nakashima, 1967; Shields et al, 1973). Some of the recently immobilized nitrogen may be mineralized fairly quickly, but over a period of time the immobilized nitrogen equilibrates with the nitrogen in soil humus, becoming progressively more resistant to mineralization in the process. Thus, although the particular nitrogen molecules which are immobilized at a given time may remain in the organic form for several years, but other nitrogen molecules, perhaps immobilized at a much earlier time. This is illustrated in Figure 3, showing concentrations of labeled fertilizer nitrogen and unlabeled soil nitrogen in the inorganic form in a soil during and after a period of rapid immobilization induced by addition of barley straw to the soil. The recently added nitrogen, applied as labeled ammonium sulfate in this case, was mostly immobilized within 15 days, with little change during the subsequent 45 days, whereas mineralization of unlabeled nitrogen proceeded during

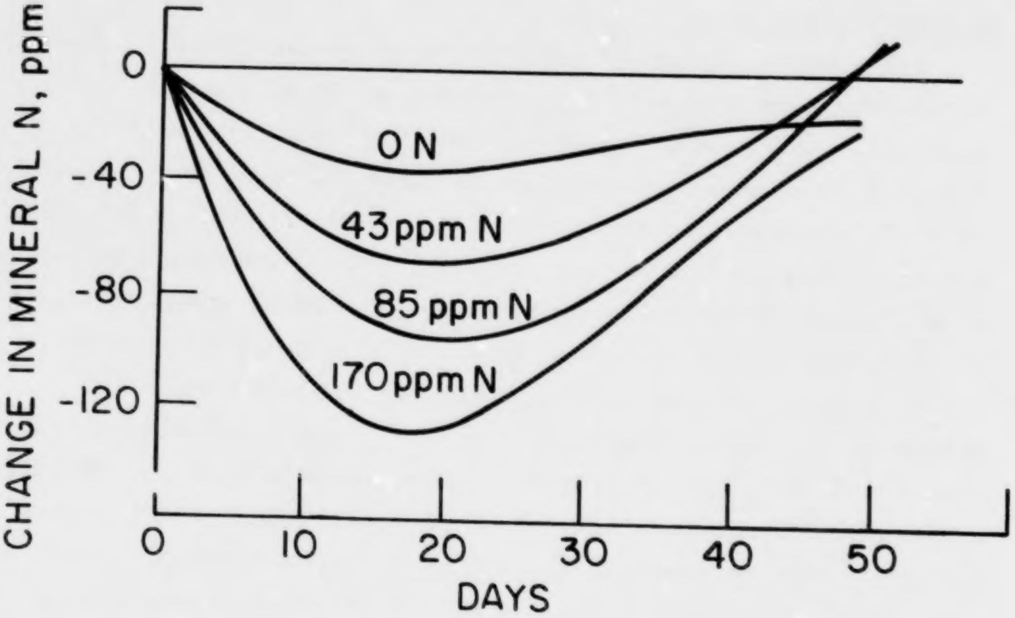

Figure 2. Effect of quantity of mineral nitrogen applied to soil on the amount of nitrogen immobilized. From data of Munson and Pesek (1958).

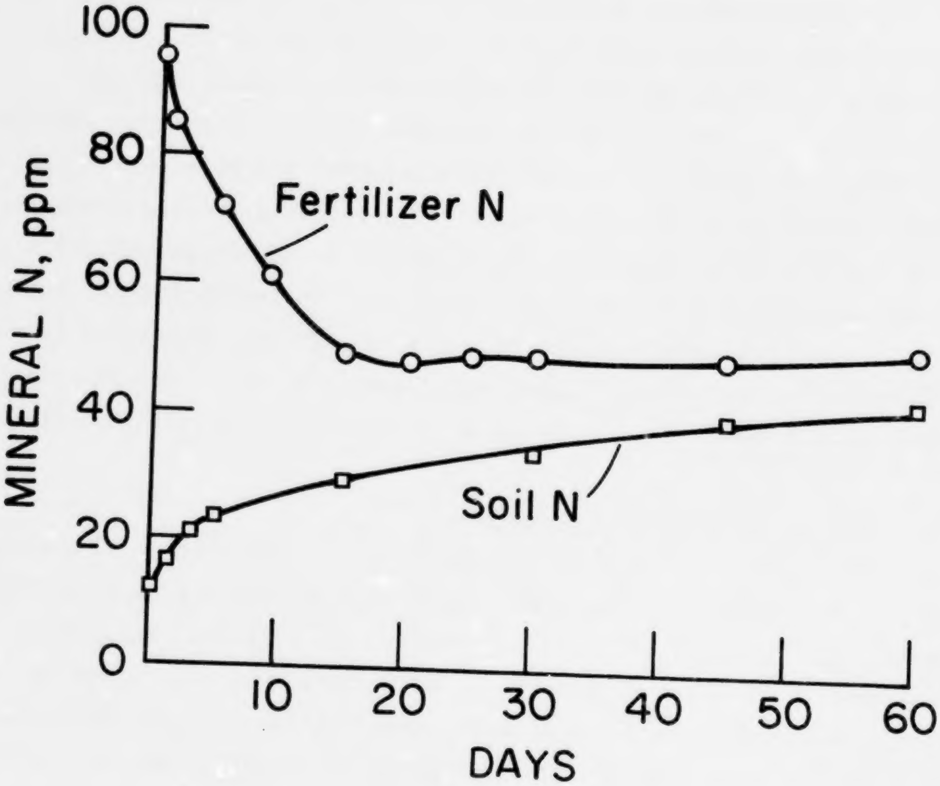

Figure 3. Illustration of mineralization of soil nitrogen during a period of immobilization of fertilizer nitrogen following addition of barley straw.

the time of maximum immobilization and continued at a lower rate throughout the period of measurement. These data illustrate the great value of tracer nitrogen in making it possible to measure opposing reactions proceeding simultaneously.

Immobilization Relative to Nitrate Pollution

Net immobilization has the effect of removing nitrate from the soil solution, and in this way may reduce the quantity of nitrogen susceptible to leaching in a situation where inorganic nitrogen surplus to the needs of crops is present. The difficulty of sustaining conditions conducive to net immobilization for more than a few weeks limits the practical value of deliberately promoting immobilization. An even more important consideration is the incompatibility of net immobilization with crop production. In the long term, buildup of nitrogen-containing organic residues in soil may contribute to nitrate pollution through higher mineralization rates, unless soil and crop management practices are adjusted to take this into account.

Prediction of Mineralization Rates

Many attempts have been made to devise a test which will accurately predict the quantity of nitrogen mineralized during a growing season. Clearly, such a test would be a useful aid to efficient fertilizer management. The most successful of such tests have been based on incubation procedures which utilize normal biological processes, for example, those of Bremner (1965), Sims et al (1967), Stanford and Smith (1972) and Stanford et al (1974). The increase in mineral nitrogen is measured over a specified period of time, usually 1-4 weeks, and the value thus obtained under controlled conditions is correlated with field responses over a growing season. The principal drawback of incubation procedures is that they are time-consuming and not well adapted to routine soil testing. Accordingly, considerable attention has been given to development of a chemical procedure which correlates well with mineralization rates estimated by the more time-consuming incubation techniques. One promising method is the 16-hour autoclaving procedure of Stanford and Smith (1976), which was compared with values obtained by an incubation procedure for 475 soils representing 54 soil types. An overall correlation coefficient of 0.87 was obtained. It appears that with the exception of certain kinds of soils, notably highly calcareous ones, this chemical procedure may be applicable to a rather broad range of soils in estimating plant-available nitrogen.

A-Value Method

Another approach to estimating the nitrogen supplying capacity of soils is the A-value concept, defined by Fried and Dean (1952) as the quantity of a given plant nutrient in soil which is equivalent in availability to that of the same nutrient added as fertilizer. In practice the fertilizer is isotopically labeled to permit differentiation between soil and fertilizer nitrogen in the crop. The amount of available nitrogen in the soil, designated by the symbol A, is given by the equation

$$A = T_s \frac{U_c}{T_c}$$

where T_s is the amount of labeled or tagged fertilizer applied, U_c is the amount of soil-derived or untagged nitrogen in the crop, and T_c is the amount of fertilizer-derived or tagged nitrogen in the crop. The equation simply states that the plant will take up soil and fertilizer nitrogen in the same proportions as they occur in the soil solution. Theoretically, the A-value is independent of the fertilizer level, but in practice it is often found to vary with fertilizer rate. It is also affected by nature of the crop, form and placement of fertilizer, and other variables (Broadbent, 1970). Although not adapted to routine determination because of the sophisticated and expensive equipment required for analysis, A-values correlate well with actual uptake of soil nitrogen, as shown in Figure 4, calculated from data of Legg and Stanford (1967) involving 12 soils fertilized at 3 rates. By contrast with the correlation coefficient of 0.97, total nitrogen in these same soils when plotted against soil nitrogen uptake had a correlation coefficient of only 0.55. One of the significant findings of recent field research with ^{15}N-depleted fertilizer is that A-values for a variety of soils and crops can be easily and inexpensively determined.

MINERALIZATION OF NITROGEN IN ORGANIC AMENDMENTS

When animal manures, sludges, or other organic wastes are applied to soils, particularly where large amounts are used, the rate of nitrogen mineralization is of importance not only in relation to the fertility status of the soil, but also in relation to the potential hazard of nitrate pollution. Because of the great variability in composition of such materials, it is difficult to make general statements regarding mineralization rates, particularly since such rates are subject to rapid change over time.

Animal Manures

Estimates of rates of mineralization of organic nitrogen in animal manures can be obtained from field trials comparing crop response at various application levels to responses obtained with different rates of inorganic fertilizers as sources of nitrogen. A comparison of this kind based on the data of Tyler et al (1964) indicated that about 41% of the nitrogen in dry steer manure was mineralized the first year and 25% the second year. Herron and Erhart (1965) compared the nitrogen supplied to sorghum by manure at three levels compared to that furnished by ammonium nitrate.

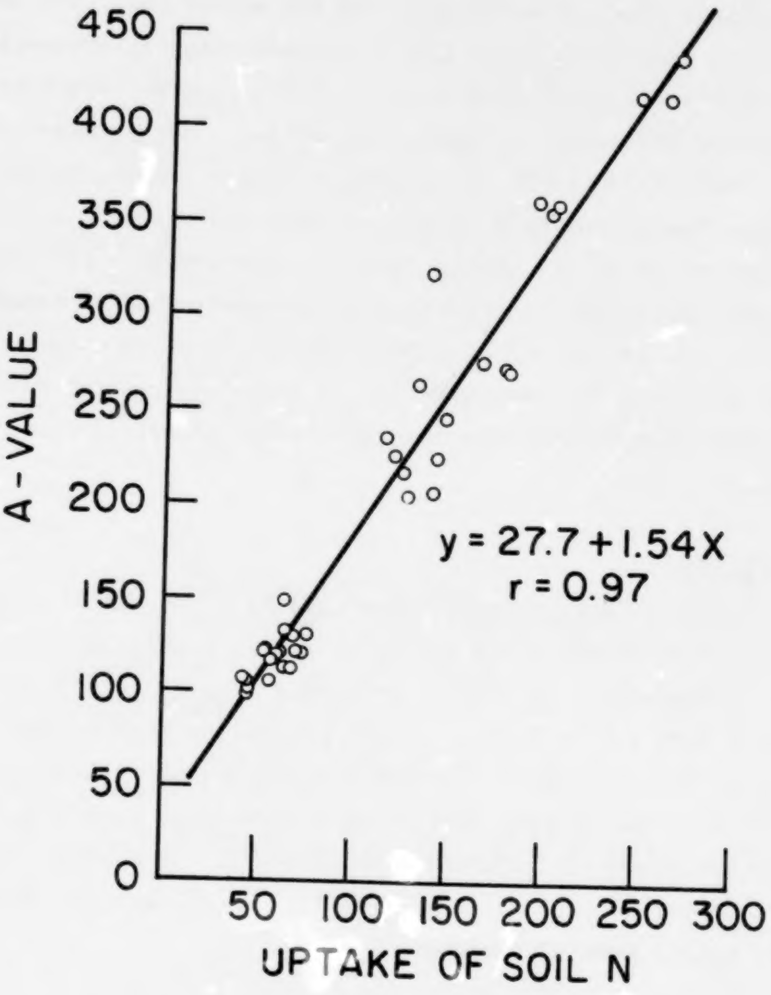

Figure 4. Correlation of A-value with uptake of soil nitrogen, calculated from data of Legg and Stanford (1967).

Gilmour et al (1977) calculated an equation from their data relating total mineralized nitrogen to elapsed time, showing 34% mineralization the first year, an additional 17% the second year, 11% the 3rd year, and so on. Pratt et al (1973) suggested the determination of nitrogen mineralization rates based on "decay series" which assume progressively decreasing decomposition rates. They stressed the need for long term field trials to estimate nitrogen mineralization rates from animal manures under a variety of soil and climatic conditions. Mineralization rates are highly variable, depending on the nature of the manure, conditions of storage prior to application, climatic conditions, and the amount of straw, sawdust or other such bedding material which may be mixed with it. Nitrogen in poultry manure is more readily mineralized than that from large animals (Macmillan et al, 1972).

Sewage Sludge

In sewage sludge, nitrogen mineralization is usually somewhat less rapid than in other organic materials of comparable nitrogen content. Peterson et al (1973) reported nitrogen contents in sludges as ranging from 3.5 to 6.4%. As much as half or more of this nitrogen may be present as ammonia. Considering the organic nitrogen in anaerobic digested sludge, Ryan et al (1973) found from 4 to 48% of the nitrogen converted to nitrate in 16 weeks. This wide range of values obtained under a variety of conditions illustrates the difficulty of making generalizations.

NITRIFICATION

The sequel to nitrogen mineralization in soils is the process whereby ammonium is oxidized to nitrate, called nitrification. The reactions may be written as follows:

$$NH_4^+ + 1\tfrac{1}{2}\, O_2 \longrightarrow NO_2^- + H_2O + 2\, H^+$$

$$NO_2^- \ \tfrac{1}{2}\, O_2 \longrightarrow NO_3^-$$

The first of these reactions is carried out by bacteria of the genera *Nitrosomonas* and *Nitrosococcus*, and the second reaction by *Nitrobacter* species. Nitrifiers are abundant in most soils except those which are rather acidic, and are capable of remaining active over a wide range of temperature and moisture conditions. These bacteria are autotrophic; that is, they require no organic matter as a source of food since they derive their energy from the oxidation of ammonium or nitrite. They do, however, require several inorganic ions including micronutrients in order to function properly. It rarely occurs that nitrification is inhibited in soil systems as a result of nutrient deficiency.

There are in soils some heterotrophic microorganisms, which require organic matter as a source of food, capable of converting ammonium to nitrite, and a few of them can produce nitrate (Alexander, 1965). Thus far, the evidence indicates that heterotrophic nitrification is of minor importance in soils.

Although normally nitrate is the principal product of the nitrification reactions, under certain conditions other compounds may be produced. One of these of particular interest is nitrous oxide, a gas which may be evolved from the soil and lost. Nitrous oxide evolution is commonly associated with denitrification, but as suggested by Yoshida and Alexander (1970) its presence in the atmosphere may be partly a result of nitrification. These investigators showed that high ammonium levels favored the formation of nitrous oxide by nitrifying bacteria.

Influence of Environmental Factors on Nitrification

Ammonia Concentration. One of the curious things about nitrifying bacteria is that they are quite sensitive to ammonia concentration, even though ammonia is their source of energy. Thus, high ammonia concentrations in the vicinity of a fertilizer band may retard nitrification. The nitrite oxidizers are particularly sensitive to free ammonia. At pH values above 7, where the equilibrium between ammonium ion and free ammonia is shifted in the direction of producing more free ammonia, there may be a temporary accumulation of nitrite in soils treated with ammonic fertilizers or urea owing to inhibition of Nitrobacter species.

pH. The optimum reaction for nitrification is in the range 7.5-8.5. Under acid conditions, the activity of nitrifying bacteria falls off sharply, ceasing altogether when the pH drops to about 4.0 (Weber and Gainey, 1962). Because the nitrification process itself generates acid, very large and rapid changes in soil pH may occur in poorly buffered soils, especially in zones of active nitrification. For example, it was shown by Broadbent et al (1957) that the pH of a sandy soil treated with aqueous ammonia dropped from 9.6 to 4.5 over a six-week period as a result of nitrification. Liming often stimulates nitrification because it neutralizes the acid produced by conversion of ammonium to nitrate. Fertilizer materials which are alkaline in reaction, such as aqueous and anhydrous ammonia, or which become alkaline after addition to soil, such as urea, tend to be nitrified more rapidly than are materials such as ammonium sulfate or ammonium nitrate which are acidic in reaction.

Temperature. Assuming that other conditions are favorable, nitrification is affected by temperature in a way similar to other biological processes. The minimum temperature is near the freezing point. Measurable oxidation of ammonium and nitrite has been reported at 2C (Frederick, 1956). The optimum temperature varies somewhat with location, since the nitrifying bacteria can adapt somewhat to their environment. The optimum temperature is higher in warm climates than in cool ones (Mahendrappa et al, 1966), but usually falls in the range between 25 and 35C. Above 40C the rate of nitrification falls off sharply, but as a practical matter, lack of moisture probably inhibits nitrification at high temperatures more than does temperature *per se*.

Oxygen. The nitrifying bacteria are obligate aerobes, but they function at oxygen concentrations somewhat lower than that of the atmosphere (Amer and Bartholomew, 1951). Nitrification is most rapid near the soil surface, but can occur at considerable depths if substrate is present.

Rates of Conversion of Fertilizer Nitrogen

In agricultural practice most of the nitrogen applied to soil is either in the ammonic form or one which is rapidly converted to the ammonic form, as in the case of urea. The availability of this nitrogen to growing plants and its susceptibility to leaching are influenced by the nitrification process, which converts the relatively immobile ammonium ion to the highly mobile nitrate. Nitrification rates are commonly considered to follow first-order kinetics; that is, the rate of production of nitrite is proportional to the amount of ammonium present at a given time, and the rate of nitrate formation is proportional to the concentration of nitrite. Normally nitrite is oxidized to nitrate as fast as it is formed, and oxidation of ammonium to nitrite is the rate-limiting step. Using a value of 25 kg N/ha/day as a reasonable estimate of maximum nitrification rate, based on rates reported by Tyler et al (1959) for a variety of soils, it can be calculated that 90% of a fertilizer application in the range of 100 kg/ha would be nitrified within 8 days at 24C. At 7C, 32 days would be required for 90% conversion.

Nitrification Inhibitors

Because nitrification converts nitrogen to a form more susceptible to loss by leaching and by denitrification, scientists have searched for many years for ways to retard or inhibit the nitrification process. Two approaches have been followed. The first is to add a toxic substance to the soil or to the ammonic fertilizer which will inhibit the nitrifying bacteria without adversely affecting plant growth or the activities of other soil microorganisms. Naturally occurring substances have been found which have these properties, for example, exudates from the roots of prairie grasses (Neal, 1969) and tannins produced from the leaves of oak and oak-pine forests (Rice and Pancholy, 1973). A few synthetic compounds of greater potency than the natural ones have been developed, and a few are being marketed commercially. One of these in particular, nitrapyrin, has been subjected to extensive testing with respect to its agronomic value, but much less is known about its usefulness in decreasing the leaching of nitrates. In many instances, improvements in yield or of nitrogen uptake efficiency have been achieved with the use of nitrification inhibitors, but as a rule, the improvements have not been dramatic. In reviewing the subject of nitrification inhibitors, Gasser (1970) stated, "More potent and more specific inhibitors are required to secure the large gains in the efficiency of nitrogen fertilizers that are needed in practical agriculture".

The second approach to control of nitrification rates is to prevent access of nitrifying bacteria to the fertilizer nitrogen. This has been done by incorporating the nitrogen into large organic molecules as with urea-formaldehyde and isobutylidene diurea formulations, thereby retarding nitrogen mineralization, or by means of coatings. Sulfur coatings are the best known example of the latter. These simply provide a physical barrier which prevents the fertilizer from going rapidly into solution. Since such preparations are more expensive than conventional fertilizers, their economic use requires improvements in fertilizer performance commensurate with their higher cost. With few exceptions, improvements of this magnitude have not been realized with any of the so-called slow release fertilizers. It may be reasoned that the value of pollution control should also be considered in weighing the evidence for and against inhibitors, coatings, or other formulations designed to achieve control of nitrification rates. However, there is as yet insufficient evidence to demonstrate that these are of real value in minimizing nitrate leaching from soils. In some instances, nitrogen uptake efficiency has been poorer with coated fertilizers than with conventional ones, which would have the effect of leaving a larger quantity of residual nitrogen subject to leaching. Gradual release of mineral nitrogen during the growth of a crop actively assimilating nitrogen may well decrease the amount of nitrate subject to leaching during the growing season, but continued release of nitrogen after the crop is removed can result in substantial leaching losses overall.

MANAGEMENT PRACTICES

There is a certain degree of inevitability about nitrogen mineralization and nitrification in soils. These processes occur in nature because they yield energy to the microorganisms responsible for them. Rates of reaction can be altered by management, but only within limits. At the present time, the best practical means available for maximizing the utilization of available nitrogen by crops and minimizing the potential for ground water pollution by nitrates produced in soil is to assure the presence of an actively absorbing root system during the period of most rapid mineralization, and to apply fertilizer nitrogen at rates such that crop needs are met but not exceeded by the supply of soil plus fertilizer nitrogen. This is illustrated in Figure 5, which presents data from a field trial with isotopically labeled nitrogen fertilizer, which made it possible to identify fertilizer-derived nitrogen in the crop and in the soil. In these experiments 180 kg of nitrogen was sufficient to produce the maximum yield of corn. Up to that level of fertilizer, there

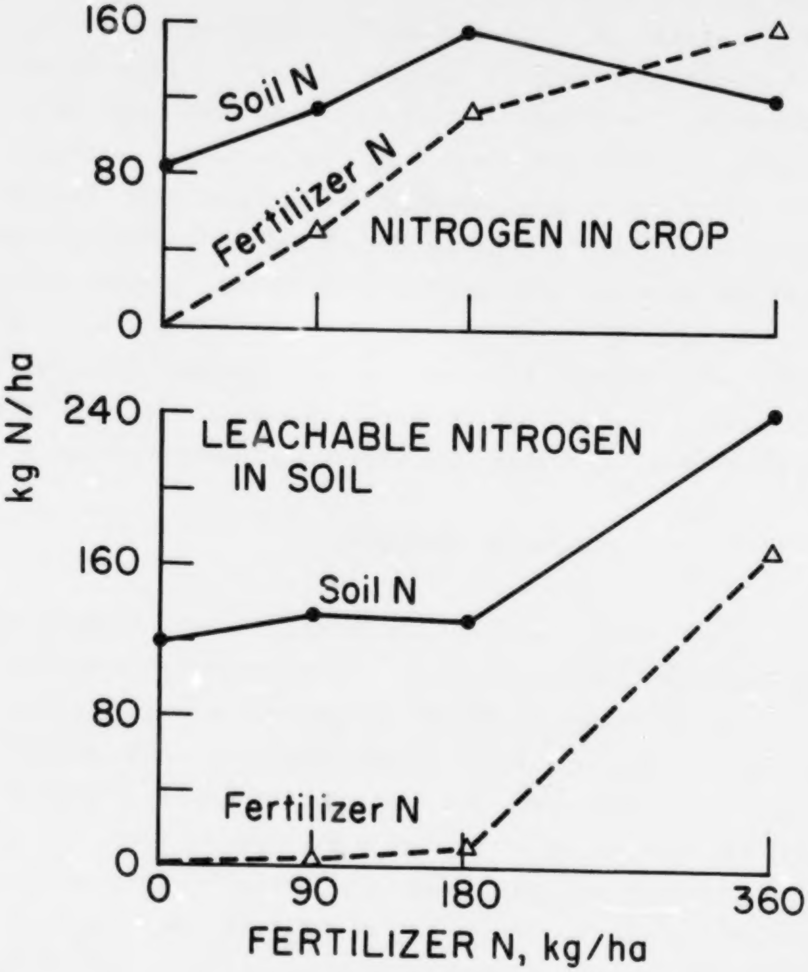

Figure 5. Influence of fertilizer rate on crop nitrogen and leachable nitrogen in soil following crop harvest.

was essentially no contribution of fertilizer nitrogen to the quantity of leachable nitrogen in the soil remaining after crop harvest, but above 180 kg the quantity of leachable nitrogen increased sharply. It will be noted that utilization of soil nitrogen was increased by fertilization up to the 180 kg rate. The data indicate that if mineralized nitrogen is fully exploited, fertilization for maximum yield is compatible with minimum pollution hazard.

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REMOVAL OF NITROGEN BY VARIOUS IRRIGATED CROPS

T. C. Tucker and R. D. Hauck[1]

ABSTRACT

Harvested agronomic and horticultural plants, excluding rice, are grown under irrigation on over 16 million hectares of land in 33 states. This paper presents results of a survey of published and recent unpublished information on removal of soil and fertilizer nitrogen from irrigated land in the harvested portion of crops. The amount of nitrogen removed varied considerably among different plant species and within the same species when grown under widely different management and environmental conditions. The amount of fertilizer nitrogen apparently removed during harvest was greater for hay, silage, and pasture crops and least for certain vegetables, fruits, and nuts. The amounts of nitrogen removed by grain crops generally was within the average range for all crops, 40 to 60% of the nitrogen applied.

Fertilizer efficiency is discussed from three main viewpoints: in terms of (1) the amount of applied nitrogen found in the plant, (2) the yield of harvested plant parts in relation to amount of fertilizer nitrogen applied, and (3) the cash value of marketable crop in relation to nitrogen cost. Examples are given, using data for coastal bermudagrass and wheat, of fertilizer efficiency as calculated on the basis of these viewpoints.

Methodological problems of collecting data and problems of data

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interpretation are discussed in relation to the several concepts of fertilizer efficiency, and emphasis is placed on the need to recognize the role of all soil, fertilizer, and crop management factors in determining the efficiency by which plants use nitrogen.

INTRODUCTION

The amount of nitrogen contained in the harvested crop generally exceeds the amount of any other nutrient. The few exceptions include such crops as potatoes, sugar beets, sugarcane, and certain vegetables. Because crop removal of nitrogen is appreciable, it is important from the viewpoints of economics, energy conservation, and environmental concerns.

In this paper, we have searched the published literature for basic information on nitrogen removal by irrigated crops and have surveyed research and extension workers for recent and unpublished data. Our intention is not to present a comprehensive review of the hundreds of citations available, but to indicate the current status of knowledge as revealed by the relatively recent literature.

LAND AREA IN PRINCIPAL CROPS

In presenting information on the distribution of irrigated agriculture in the United States, we found it convenient to use the crop and state groupings used in the most recent Census of Agriculture (1974 State Summary Data). For example, all vegetable crops are grouped together, except for dry beans, peas, and lima beans, which are placed into a separate group, and potatoes, listed separately.

We have not included data for rice in the figure for land area under irrigation because rice irrigation practices differ greatly from those used with all other crops. The total land area planted to rice in 1974 was slightly more than 1 million hectares, most of which was in four states--Arkansas, Louisiana, Texas, and California.

Irrigation to some extent is practiced in every state in the United States. However, in four states--Alaska, Rhode Island, Vermont, and West Virginia--the area under irrigation did not exceed 400 hectares in 1974 (1974 Census of Agriculture). Thirty-three states each had 10,000 or more hectares under irrigation, totaling over 16 million hectares. The states with the most land area under irrigation were California, Texas, Nebraska, Idaho, Colorado, and Kansas (Figure 1). The remaining 11 western states, Florida, and Oklahoma each had more than 200,000 irrigated hectares.

Except for the vast area of dryland wheat in 5 of the 11 western states in the Mountain and Pacific Divisions, the major portion of field crops in these divisions are grown under irrigation (Table 1). The relatively high percentage of total cropland under irrigation in the West South Central Division is due largely to Texas and, to some extent, Oklahoma. Most of the irrigated corn and sorghum of the West North Central Division is grown in Nebraska and Kansas, and all irrigated cotton in this division is grown in Missouri. Except for potatoes and vegetables, only small amounts of other crops are grown under irrigation in the remaining five divisions. Over one half of the land nationwide cropped to potatoes and vegetables is irrigated.

The percentage of harvested cropland that was irrigated in 1974 is illustrated by ranges in Figure 2. Of the 11 western states, 7 have more than one half of their cropland under irrigation. Large land areas of dryland wheat account for the smaller percentages of irrigated cropland in the remaining four states--Colorado, Oregon, Washington, and Montana.

The principal field crops (separately or in groups) and fruits and nuts for states with more than 10,000 hectares of irrigated land are given in Tables 2 and 3. As mentioned previously, the groupings have been made according to Census Bureau categories; in some cases,

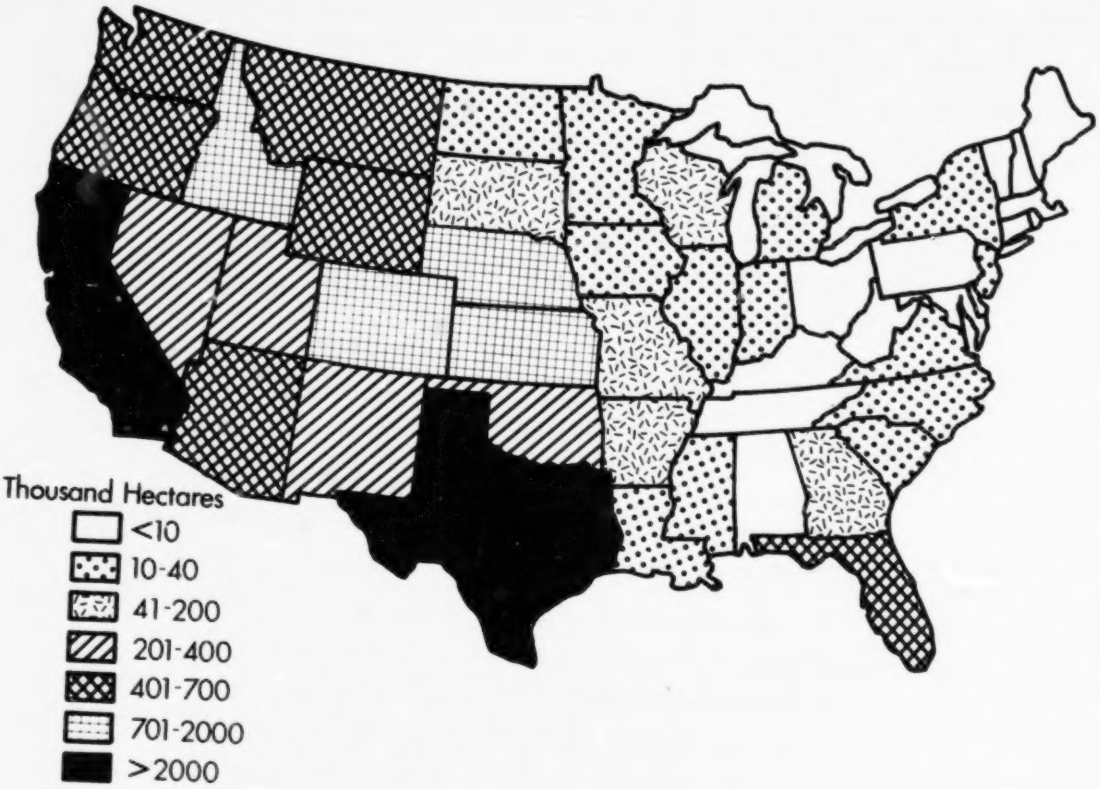
Thousand Hectares
<10
10-40
41-200
201-400
401-700
701-2000
>2000

Figure 1. Irrigated land area in the U.S. by states, 1974.

Table 1. Land area of irrigated crops as a percentage of total for each crop[1].

	All field corn	All sorghum	Wheat	Hay	Cotton	Potatoes	Vegetables
	----------------------------	----------	----------	% irrigated	----------	----------	----------------------------
U.S.	9	18	5	15	30	58	52
Divisions[2]							
New England (6)						4	23
Middle Atlantic (3)						36	28
East North Central (5)						50	15
West North Central (7)	13	7	1	3	5	11	4
South Atlantic (8)	1	1	1	1		38	52
East South Central (4)					1	8	10
West South Central (4)	63	25	10	7	29	74	65
Mountain (8)	94	46	10	69	100	99	94
Pacific (5)	92	96	16	68	100	98	87

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[1] Calculated from 1974 Census of Agriculture state summary data. Values are rounded to nearest 1%. Individual states are given on pages II-33 to II-44.

[2] Number of states in each division given in parenthesis.

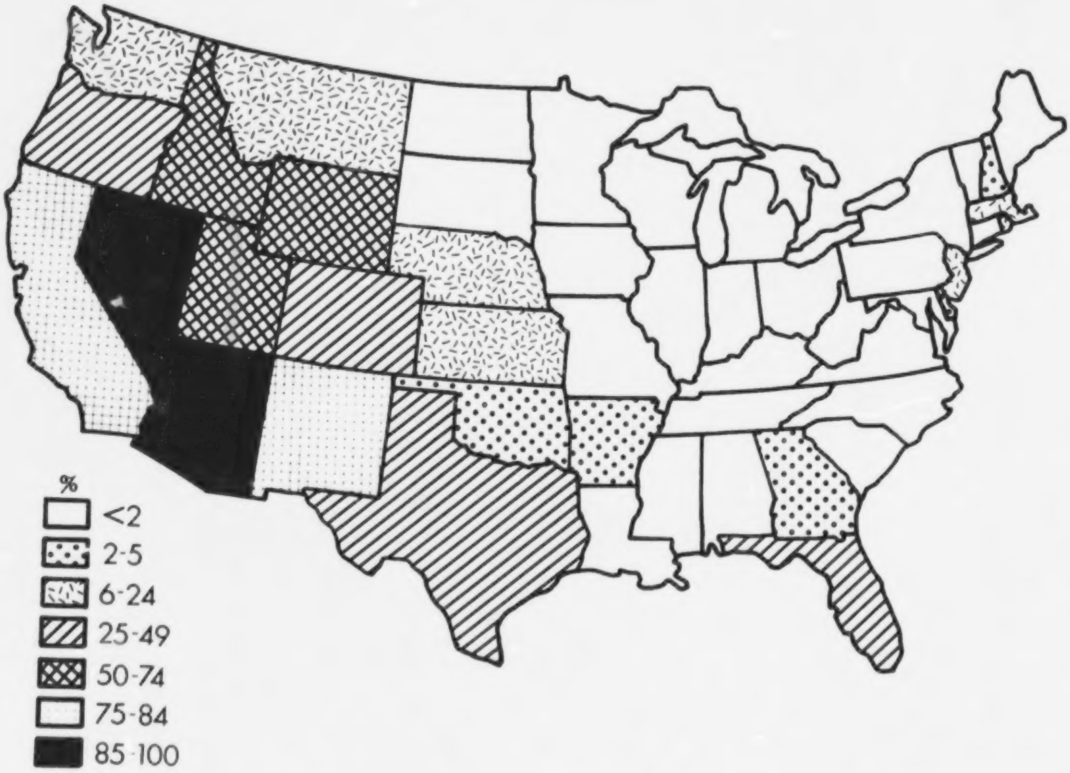
%
<2
2-5
6-24
25-49
50-74
75-84
85-100

Figure 2. Percentage of harvested cropland under irrigation in the U.S. by states, 1974.

Table 2. Irrigated land area in selected field crops (ha x 10^3), 1974.

	Total irrigated	All field corn	Corn grain	All sorghum except syrup	Wheat grain	All hay except sorghum	Peanuts	Cotton	Potatoes	Vegetables	Soybeans	Barley	Sugar beets	Dry beans, peas, lima beans	Oats
U.S.	-	2,703	2,239	1,073	1,311	3,184	72	1,498	311	648	192	543	310	262	71
CA	2,898	140	77	61	151	485	-	465	24	298	-	162	95	77	7
TX	2,414	225	199	650	357	96	36	733	6	44	33	10	7	3	5
NE	1,603	1,259	1,183	51	15	128	-	-	4	-	36	1	29	38	4
ID	1,134	45	13	< 1	177	345	-	-	126	15	-	121	36	69	7
CO	1,129	294	205	34	51	398	-	-	18	9	-	61	48	30	7
KS	813	421	375	134	155	54	-	-	-	< 1	8	< 1	13	2	< 1
MT	696	25	2	-	27	412	-	-	2	-	-	54	16	3	10
FL	624	8	7	< 1	-	5	< 1	-	8	75	8	-	-	-	-
OR	611	13	2	-	50	244	-	-	22	47	-	24	5	2	9
WY	578	31	10	-	4	320	-	-	2	-	-	34	21	9	10
WA	518	19	7	-	121	123	-	-	36	29	-	5	25	18	1
AZ	426	5	< 1	40	69	74	-	160	4	21	-	20	3	< 1	< 1
UT	363	28	4	< 1	22	177	-	-	2	3	-	37	6	-	3
NM	328	23	15	60	36	80	2	58	1	6	-	6	-	2	1
NV	298	1	-	-	11	170	-	< 1	3	-	-	4	-	-	< 1
OK	206	34	27	36	48	25	20	24	< 1	< 1	2	< 1	-	-	< 1
AR	99	< 1	< 1	2	< 1	2	-	33	-	2	54	-	-	-	< 1

(Continued on next page)

Table 2. Continued

	Total irrigated	All field corn	Corn grain	All sorghum except syrup	Wheat grain	All hay except sorghum	Peanuts	Cotton	Potatoes	Vegetables	Soybeans	Barley	Sugar beets	Dry beans peas lima beans	Oats
SD	61	26	18	< 1	2	24	-	-	-	-	1	< 1	-	< 1	1
MO	57	18	17	3	2	1	6	-	-	< 1	24	-	-	-	-
WI	51	10	9	-	-	1	-	-	14	18	-	-	-	< 1	< 1
GA	45	15	13	-	-	< 1	13	< 1	-	3	1	-	-	-	-
MI	38	9	8	-	-	< 1	-	-	6	12	< 1	-	-	1	-
NJ	36	-	-	-	-	-	-	-	3	23	1	-	-	-	-
MN	31	11	10	-	< 1	2	-	-	7	1	2	-	-	2	-
ND	28	4	< 1	-	4	11	-	-	< 1	-	-	< 1	4	2	< 1
NY	22	< 1	-	-	-	-	-	-	9	7	-	-	-	-	-
IL	22	9	8	-	-	-	-	-	< 1	4	4	-	-	-	-
LA	20	< 1	-	-	-	< 1	-	5	-	< 1	7	-	-	-	-
NC	20	1	1	-	-	-	-	-	-	< 1	< 1	-	-	-	-
IA	16	13	12	-	-	-	-	-	-	-	1	-	-	-	-
MS	16	< 1	-	-	-	-	10	-	-	-	2	-	-	-	-
IN	13	5	5	-	-	-	-	-	2	2	1	-	-	-	-
VA	11	< 1	< 1	-	-	-	-	-	2	2	-	-	-	-	-

Table 3. Irrigated land area in selected fruit and nut crops (ha x 10^3), 1974.

	Apples	Peaches	Pears	Cherries	Grapes	Plums and prunes	Valencia oranges	Navel oranges	Other oranges	Grape-fruit	Lemons	Other citrus	Stone-fruit	Almonds	Other nuts
U.S.	56	41	31	18	255	49	164	50	156	92	32	16	54	111	106
CA	8.9	36.4	15.0	5.3	246	43.7	30.4	41.7	0.8	8.5	21.5	2.0	51.4	111	72.9
TX	-	-	-	-	-	-	4.9	0.8	6.5	18.2	-	-	-	-	18.2
ID	2.0	0.4	0.4	-	-	0.8	-	-	-	-	-	-	-	-	-
CO	2.4	0.8	-	-	-	-	-	-	-	-	-	-	-	-	-
FL	-	-	-	-	-	-	123.5	5.3	147.4	60.7	2.8	12.6	-	-	-
OR	2.4	0.8	8.1	6.9	-	3.2	-	-	-	-	-	-	-	-	8.9
WA	38.5	1.2	7.7	4.5	7.3	0.8	-	-	-	-	-	-	-	-	-
AZ	-	-	-	-	1.6	-	5.7	2.0	0.8	4.0	7.7	1.6	-	-	5.7
UT	1.2	0.8	-	1.2	-	-	-	-	-	-	-	-	-	-	-
NM	0.8	-	-	-	-	-	-	-	-	-	-	-	-	-	-

small acreages of crops were combined. States are listed in descending order based on total irrigated land area. Slight differences may be noted between the sum of land areas for a particular crop and the total for the United States because data for some very small areas of irrigated land were omitted and individual values were rounded to the nearest thousand hectares.

Only a few states are involved in the production of certain irrigated field crops such as peanuts, dry peas, lima beans, tobacco, fruits, and nuts. Most of the irrigated land (> 75% of the United States total) in corn, sorghum, wheat, cotton, potatoes, vegetables, soybeans, barley, sugar beets, and dry beans is located in only six states. A major position of the total irrigated acreages of a particular crop may be grown in a single state (e.g., Nebraska, corn; Texas, sorghum; and Idaho, potatoes).

Land areas of crops under irrigation have remained constant or declined slightly in some states because of limiting water supplies. In other states irrigation of cropland has increased from a small to a marked amount since the 1974 census (e.g., Nebraska has more than doubled its land area under irrigation since 1974).

Census data show that irrigated acreages are increasing in many states in regions which previously had little or no irrigated agriculture. Although as yet confined to a small percentage of cropland in these states, irrigation is being practiced to provide supplemental water. This indicates that management practices common to irrigated agriculture are becoming more important nationwide. The increase in yields obtained from irrigation during otherwise moisture-deficient periods in turn leads to increased removal of nitrogen in the harvested crop. Adoption of irrigation practices requires changes in fertilizer and other management practices, and it is obvious that high yield potentials are achieved only through efficient management of both fertilizer nitrogen and water.

Data on average crop yields, nitrogen concentration, and irrigated acreage permit one to calculate nitrogen removal in the harvested portion of each crop by state or region. Garman and White (1964) reported such information for several crops. They used values for percent nitrogen in various crops which are similar to those being reported currently for the same crops grown under irrigated conditions, but their yields and, consequently, calculations of total nitrogen removed are somewhat lower than those reported elsewhere for most irrigated crops. Nevertheless, calculations such as those reported by Garman and White can be used in part to estimate future nitrogen needs based on total acreages of each crop.

SOIL-FERTILIZER NITROGEN AND CROP REMOVAL

For a particular plant species, a relative index of the effectiveness by which plants use nitrogen can be obtained from noting the amount of nitrogen removed in the harvested portion of the crop in relation to the amount of fertilizer nitrogen applied. This relative index can be used in comparing nitrogen removal by the same crop at different locations or under different crop and fertilizer management systems. However, the relative index should be used with caution to compare nitrogen removal among different plant species because the total nitrogen taken up may be distributed between the harvested and nonharvested portions of the crop in an entirely different way in one plant species as compared to another. For most crops of concern here, a large proportion of the total nitrogen in the plant is removed from the land during harvest, but the total amount of nitrogen removed may vary considerably within the same plant species (depending on site and management factors) and among different species. For example, wheat grown at either of two locations was found to contain about 75% of its total nitrogen in the grain. However, because of differences in grain yields, the amount of nitrogen removed from each site was different. For cotton, only about 50% of the nitrogen taken up is removed in the harvest portions of the plant, and the total amount removed may be very different from that removed by wheat.

Cotton and wheat remove different amounts of nitrogen from the land because they differ in their dry matter yields, the nitrogen concentrations in their various plant parts, and in the proportion of total plant nitrogen which is removed at harvest.

In the discussion which follows, let us assume that the nitrogen removed from the land at harvest is derived solely from soil and fertilizer. For nonleguminous plants, we include in soil nitrogen the relatively small amount of nitrogen added during the season through nonsymbiotic biological nitrogen fixation.

Nonleguminous plants take up nitrogen almost entirely in the ammonium and/or nitrate form. Under aerobic soil conditions and with other factors favorable, soil- and/or fertilizer-derived ammonium at low concentrations in soil is rapidly changed to nitrate (via the biological nitrification reaction). Under these conditions, plants normally take up most of their nitrogen as nitrate. However, in concentrated bands of ammonium or ammonium-producing fertilizers (e.g., urea), nitrification proceeds more slowly and, consequently, appreciable amounts of ammonium may be taken up by the plant. Whether nitrogen in the soil-plant system is in the ammonium or nitrate form may determine the amount of nitrogen in the plant and the manner in which it is used. The plant physiological aspects of ammonium versus nitrate nutrition will not be discussed here, but we will be concerned with the form of nitrogen as it affects nitrogen supply to the plant. Nitrification converts a relatively immobile form of nitrogen, ammonium, to a relatively mobile form, nitrate, and produces oxidized forms of nitrogen, nitrite and nitrate, which are subject to loss through leaching and denitrification.

Ideally, we would like to relate crop removal of nitrogen from the land to the overall nitrogen balance of the soil-fertilizer-plant system. For a first approximation of a working model of such a system, let us assume that the crop can remove all (100%) of the available forms of nitrogen added, already present, and to be formed in the root system sorption zone (Bray, 1954). Let us also assume no nitrogen losses from the system but a soil nitrogen supply during the growth of the crop that is equal only to that which is returned to the land as crop residue. Obviously, the soil nitrogen supply is deficient and optimum yields cannot be obtained. In this highly stylized and simple model, the nitrogen fertilizer requirement becomes equal to the amount of nitrogen removed in the harvest portion of the plants. It follows that the soil nitrogen requirement (SNR) is equal to the amount of nitrogen removed by the crop plus the amount returned to the land. The fertilizer nitrogen requirement (FNR) is equal to the soil nitrogen requirement minus the amount (or level) of nitrogen in the soil (SNL), that is, FNR = SNR - SNL. The amount of available nitrogen either present, added, or formed in the root zone of the soil must be at least equal to the total crop content of nitrogen at the yield level acceptable for that crop. Realizing the tremendous over-simplification of this model, we present it nevertheless to serve as a starting point for modeling a more realistic system, and to dramatize the importance of considering the amounts of nitrogen removed in the harvested parts of crop plants.

In practice, the values for SNR and FNR must be multipled by a factor which considers nitrogen losses by leaching and volatilization, nitrogen removal from the plant-available form by immobilization reactions in soil, movement of nitrogen from the plant root system, and incomplete uptake and use of nitrogen because of environmental stresses. Gains of nitrogen from rainfall and biological nitrogen fixation also must be considered. Obviously, accurately determining this factor presents many problems.

NITROGEN CONTENT OF CROPS

Total Uptake

The total amount of nitrogen in the plant is equal to the amount of total dry matter times its average nitrogen concentration (percentage). As discussed in the preceding section, this value represents the minimum soil nitrogen requirement, that is, the supply of available nitrogen from the soil plus fertilizer nitrogen must equal the total amount in the plant, assuming no nitrogen losses and complete recovery of available nitrogen by the plant. Because losses of plant-available nitrogen occur and the plant does not take up all available nitrogen, nitrogen supplies greater than the minimum SNR must be present in order to obtain optimum yields; how much greater depends on the crop. For example, for lettuce, carrots, and hay crops, almost all of the nitrogen taken up is removed from the land. Crop removal of nitrogen is almost equal to the minimum SNR. On the other hand, crops grown for grain return appreciable amounts of nutrients to the soil and removal of nitrogen in the harvested portion of the crop is less than the minimum SNR. In either case, the amount of soil and fertilizer nitrogen needed is determined both by the total amount of nitrogen removed and the crop and soil factors which determine how much nitrogen is taken up by the plant.

Crop Removal

Our survey of data from research and extension workers showed a very great variation in yields and nitrogen contents among different plant species as well as within the same species. Rather than present average or mean values, the range of values obtained is given in Table 4, except for reports of single values. Usually, single values for yield and nitrogen content were taken from a California study (Rauschkolb and Mikkelsen, 1978). In many instances, average farm applications of nitrogen exceeded the rates usually applied at experimental fields at a

Table 4. Summary of survey data for yield, nitrogen removal, and fertilizer application under irrigation.

Crop	Yield	Total nitrogen content of: Harvested portion	Total nitrogen content of: Total crop	Fertilizer nitrogen applied	Average farm application
Agronomic crops			kg/ha		
Cotton	900- 1,700	63-109	125-230	100-134	90-109
Corn	7,200-10,000	110-180	165-280	160-280	160-280
Barley	3,760- 4,500	72- 85	85-125	140	90
Oats	3,000- 3,600	63- 67	94-100	72-140	72- 90
Silage, Corn	6,000-29,000	111-361	-	150	240
Sorghum	4,500- 7,800	84-157	137-235	123-224	110-224
Wheat	2,700- 7,700	56-172	72-262	56-224	45-224
Beans, dry	3,000- 5,800	90-116	168-184	56	56
Sugar beets	56,000-86,300	112-167	168-253	224-336	250-440
Hay, all non-legume	4,500-18,000	94-438	-	70-700	120-250
Safflower	3,000	92	120	-	-
Vegetables					
Asparagus	3,800	17	-	111-209	159
Beans, lima	3,360	84	-	168	67
snap	12,000	52	-	60-125	86
Broccoli	11,530	60	-	-	280
Cabbage	30,000	67	-	106-194	147
Cantaloupe	22,400	36	-	77-171	106
Carrots, roots	42,000	76	-	87-164	134
tops	20,000	98	-	-	-
Cauliflower	15,000	56	-	142-263	205
Celery	75,000	135	-	222-458	321
Cucumbers	30,000	30	-	102-194	136
Lettuce	30,000	48	-	130-228	178

(Continued on next page)

Table 4. Continued

Crop	Yield	Total nitrogen content of: Harvested portion	Total crop	Fertilizer nitrogen applied	Average farm application
Vegetables (continued)					
Onions	36,000	86	-	125-224	163
Peas, green	3,600	36	-	48- 97	59
Peppers, green	17,233	113	68-120	143-256	252
Potatoes	34-45,000	227-229	281-317	276-356	400-500
Spinach	22,400	114	-	142-264	196
Sweet corn	12,000	77	-	116-222	162
Tomatoes	56,000	101	-	103-212	159
Fruits and nuts*					
Apples	36,000	29	94	41-134	92
Apricots	18,000	34	114	67-148	96
Avocados	5,800	20	-	92-235	140
Cherries	9,000	19	-	40- 99	73
Citrus:					
Oranges	18,000-55,000	24-90	-	95-217	138
Grapefruit	24,000-48,000	38-78	-	124-280	172
Lemons	30,000-73,750	57-92	-	132-270	186
Tangerines	12,000	16	-	118-243	159
Figs	6,500	12	-	54-122	92
Grapes	22,400	22	82	32-111	63
Olives	4,600	9	-	46-150	92
Peaches	36,000	50	-	87-198	144
Pears	34,000	25	-	74-194	133
Prunes and plums	18,000	54	-	52-156	115
Strawberries	44,000	66	-	110-246	178
Nuts:					
Almonds	2,000	60	-	81-208	142
Pecans	2,800	67	-	-	-
Walnuts	2,200	52	-	74-211	134

*50-80 kg/ha in leaves and wood recycled in fruits and nuts.

particular locality. This means that on farms where application rates were higher and yields comparable to or lower than those of experiment fields, recovery of applied nitrogen by the crop was less on farms than on experiment fields. From the data given in constructing Table 4, we calculated the percent of applied fertilizer recovered in the harvested portion of the crop, based on the lower range of values for nitrogen application rates and yields. Apparent recovery values ranged from 48 to 88% for agronomic crops.

The data in Table 5 were adapted from Miller and Smith (1976). Their values for apparent removal of applied nitrogen agree fairly well with our survey data (Table 4) for a number of crops but differ considerably for others. For example, their value for the percent of applied fertilizer nitrogen removed in corn grain (36%) is considerably lower than values obtained in our survey (average of 65%), and is in between the range of 0 to 58%, depending on rate and time of nitrogen fertilizer application, reported by Perry and Olson (1975). Similar differences in nitrogen recovery values occur for cotton. Miller and Smith (1976) reported 43%, Halvey (1976) reported 98 to 109%, and our survey showed 63 to 81%.

Two possible explanations for the above discrepancies come to mind. First, where low recoveries of applied nitrogen are obtained, the plant may be taking up a considerable amount of soil nitrogen that is assumed to be fertilizer nitrogen. (The measurement of soil versus fertilizer nitrogen is discussed later). This is especially apparent where the harvested portion of the crop contains as much or more nitrogen as was applied. Second, percent recovery of applied nitrogen is affected by level of soil nitrogen and rate of fertilizer application, among other factors. Corn and sorghum data of Perry and Olson (1975) and wheat data of Gardner and Jackson (1978) show that percent recovery of applied nitrogen decreases with increase in nitrogen application rate. Further discussion of nitrogen recovery will be given later in the section dealing with nitrogen efficiency.

Table 5. Estimated crop yields, nitrogen fertilizer application and utilization in the San Joaquin Valley of California, 1971*.

	Nitrogen applied kg/ha	Yield mt/ha**	% Removal of applied
Cereal hay	72	7.6	97
Sugar beets	127	42.5	86
Safflower	103	3.2	85
Silage	174	44.2	76
Irrigated pasture	80	9.0	68
Rice	90	4.9	68
Grain sorghum	117	4.2	64
Wheat	120	3.4	60
Barley	114	3.3	53
Cotton	115	2.1	43
Corn	242	6.3	36
Tomatoes	138	50.8	66
Carrots	133	41.7	56
Potatoes	250	39.5	53
Onions	222	51.78	50
Lettuce	222	24.1	21
Cantaloupes	134	16.3	13
Watermelons	178	28.7	12
Olives	28	5.6	50
Oranges	111	12.1	23
Peaches	154	20.2	22
Nectarines	147	15.6	18
Plums	125	11.3	7
Walnuts	141	1.6	12
Almonds	151	1.0	10
Grapes	49	14.6	37

*Adapted from Miller and Smith (1976).
**Metric tons/ha.

Data for coastal bermudagrass adapted from Fisher and Caldwell (1958) and presented in Table 6 show increasing dry matter production with increasing nitrogen application to 1,344 kg nitrogen/ha, after which yield declined.

Table 6. Coastal bermudagrass yield, nitrogen percentage, and nitrogen fertilizer efficiency calculated on the basis of three concepts*.

Applied nitrogen	Yield	Nitrogen in hay	Incremental Removal of fertilizer nitrogen	Incremental Dry matter increase	Incremental Economic return**	Accumulative fertilizer removal
kg/ha	metric tons/ ha	%	%	kg/kg nitrogen	$ output/ $ nitrogen cost	%
0	5.98	1.28	-	-	-	-
112	9.81	1.46	60	34.2	4.37	60
224	13.28	1.68	71	31.0	3.97	65
448	19.24	1.87	61	26.6	3.40	63
672	23.86	1.98	50	20.6	2.63	59
896	27.10	2.06	38	14.5	1.85	54
1,120	29.01	2.11	24	8.5	1.09	48
1,344	29.57	2.19	16	2.5	0.32	42
1,568	28.74	2.26	1	-3.7	-0.47	37
1,792	26.59	2.35	-	-9.5	-1.22	31
2,016	23.09	2.51	-	-15.6	-2.00	25

*Calculations from data of Fisher and Caldwell (1958).

**Based on hay at $0.055/kg and nitrogen at $0.43/kg.

Nitrogen concentration, on the other hand, increased from 1.28 to 2.51% at application rates of 0 to 2,016 kg/ha. Maximum nitrogen removal occurred at an application rate of 1,568 kg nitrogen/ha. At higher rates of application, increasing nitrogen concentration did not compensate for declining yield; thus, total crop removal of nitrogen also declined.

Macy (1936) hypothesized that each plant species must maintain at least a minimum nutrient (nitrogen) concentration in order to survive, this concentration being different for different species. He further stated that when the supply of nitrogen sufficient to maintain this low concentration within the plant is absorbed by and translocated from the roots in small increments, the intial effect will be to increase growth (dry matter production) without change in the plant's average nitrogen concentration. When the supply of nitrogen is greater than that needed to maintain the minimum concentration, the growth rate increases with increasing nitrogen supply and the nitrogen concentration in the plant also will increase. When maximum growth has been obtained, increasing the nitrogen supply to the roots serves only to increase the nitrogen concentration within the plant (luxury consumption). The concentration will continue to increase beyond the point where further increases in soil supply of nitrogen result in yield depression. Finally, when yield is severely depressed, nitrogen concentration decreases. Thus, at maximum yield, the nitrogen concentration within the plant is not at maximum. It reaches maximum only when yield is severely suppressed because of adverse effects on growth from oversupply of nitrogen.

The above sequence of changes cannot be illustrated by data from most experiments reported in the literature because of the relatively narrow range of nitrogen supply encountered in these experiments. It is seldom possible to show the low level effect on concentration because most cultivated soils will provide more than enough nitrogen for the nitrogen concentration in the plant to be at the survival level. In practice, a maximum concentration may be observed where fertilizer nitrogen added in excess of need for maximum production produced some yield depression (e.g., see data for coastal bermudagrass, Table 6, which shows a continuing increase in concentration with increasing nitrogen supply including the rate where yield is depressed).

FERTILIZER EFFICIENCY

Fertilizer efficiency can be defined in various ways. It can be expressed in terms of the amount of nutrient taken up by the plant and removed at harvest, the amount of nutrient used by the plant to increase growth, the economic return for amount of nutrient applied, or variations of the above. Most of the data in the preceding sections are presented in terms of nutrient uptake and/or yield. In this section we discuss fertilizer efficiency from several viewpoints.

Fertilizer Uptake Versus Use

Probably the most common expression of nitrogen fertilizer efficiency relates the amount of total nitrogen in the plant derived from fertilizer to the amount of fertilizer nitrogen applied, that is, percent recovery of applied nitrogen. Assume for the moment that the amount of fertilizer-derived nitrogen in the plant can readily be measured. Consider two nitrogen fertilizers, A and B. At equal rates of nitrogen application, if more nitrogen from A is found in the plant than from B, then A is said to be more efficient than B, using the above definition of fertilizer efficiency. However, suppose that crop yields from A and B are the same, such that even though less nitrogen was taken up from B than from A, the nitrogen from B was used more efficiently by the plant; that is, less nitrogen was needed to produce the same yield. From a plant physiological point of view, fertilizer B could be judged more efficient. It is not uncommon for nitrogen to be taken up by a plant in amounts exceeding those needed at certain growth stages, which results in some luxury consumption of nitrogen; that is, some nitrogen is not used in the production of plant cells or cell constituents.

Seldom is the nitrogen in the total plant determined. Most commonly it is the nitrogen in the above-ground portions of the plant. The agricultural economist is concerned with those parts of the plant which have marketable value. Except where nitrogen content affects crop quality, and only where crop quality has marketable value, the economist is not concerned with the amount of nitrogen in the plant derived from fertilizer, but is concerned with the yield of commodity for each increment of fertilizer applied. The economist views fertilizer efficiency in terms of the cash value of product in relation to the cost of the nitrogen input, or profit per increment of applied nitrogen.

Short-term Versus Long-term Efficiency

From the above, it can be seen that nitrogen fertilizer efficiency can be expressed in terms of: (1) the amount or percentage of applied nitrogen found in the entire plant or in the above-ground parts of the plant, (2) the yield of harvested plant parts in relation to the amount of nitrogen applied, and (3) the value of the harvested plant parts in relation to the cost of the nitrogen fertilizer application. These expressions of efficiency usually are used in relation to nitrogen uptake and/or yield during the season of fertilizer application.

A longer-term viewpoint of fertilizer efficiency considers also the residual value of the fertilizer. Commonly 20 to 45% of applied fertilizer nitrogen remains in the soil after the first growth season, and some part of this residual nitrogen is recovered by plants in succeeding years. Very little information is available on the long-term efficiency of nitrogen fertilizers because such information is difficult to obtain, requiring expensive, long-term nitrogen tracer studies in order to obtain accurate data. Whether fertilizer A is more efficient overall than B is of concern more to those interested in the total environmental impact of nitrogen fertilizers than to those who are concerned mainly with the crop production value of the fertilizers.

Measurement of Efficiency

Regardless of how one views nitrogen fertilizer efficiency, certain measurements must be made which provide the data needed to calculate efficiency. The collection of accurate yield data is essential for all calculations of efficiency. Further, it is necessary to estimate the increase in yield from fertilizer above that which would have been obtained in the absence of fertilizer. It should be noted that the yield increase is the cumulative effect of the fertilizer application on growth, including the effect of the fertilizer nitrogen on the possible increased supply of soil nitrogen to the plant and the increased ability of the plant to obtain more nitrogen and use it efficiently.

Without use of tracer nitrogen--to be discussed later--two experimental plots are needed to obtain data which will permit one to estimate fertilizer efficiency in relation to yield. However, it is desirable to have at least three rates of nitrogen application in addition to the untreated control plots, plus adequate replication, depending on site variability. One can expect yield data from a properly executed field experiment to vary by no more than 10%.

Accurately determining the amount of fertilizer-derived nitrogen in the whole plant or plant parts is more difficult than determining yield alone. The total amount of soil plus fertilizer nitrogen taken up is equal to the yield multipled by the average total nitrogen concentration in the plant(s). Since the analytical error for determining total nitrogen in plant tissue is relatively low (about 2 to 6%), the accuracy of the yield data determines the accuracy by which the total nitrogen content of the crop can be determined.

However, there is an additional source of error in the determination of the proportion of the total nitrogen in the plant which is derived from fertilizer. We refer to error in data interpretation as follows:

The difference method commonly is used in determining the amount of nitrogen in the plant derived from fertilizer (except where nitrogen tracers are used). That is, the amount of fertilizer nitrogen taken up by the plant is calculated as the difference between total nitrogen uptake from fertilized and unfertilized plots. Use of this method for calculating fertilizer recovery requires one to assume that all of the nitrogen transformations which affect the amount of soil nitrogen which is taken up by the plant occur in the same manner in both fertilized and unfertilized plots. Very often this is not a valid assumption; the nitrogen fertilizer may produce effects in the soil or on the plant's ability to obtain soil nitrogen that does not occur in the unfertilized plot. For example, the fertilizer may stimulate root growth which results in greater root exploration for soil nitrogen. Usually, the difference method gives higher values for percent recovery by plants of applied nitrogen than methods which use nitrogen tracers.

In the usual tracer method, fertilizer labeled with the stable (nonradioactive) isotopes of nitrogen, ^{14}N and ^{15}N, are applied to the soil. No control plots are necessary in the usual experiment, although some researchers have included control plots to enable them to compare the direct tracer method with the indirect nontracer method. At harvest, the tracer content of the crop is determined and the amount of fertilizer versus soil nitrogen taken up can be calculated. Because the tracer method is not dependent upon use of data obtained from a control plot, and because measurements are made directly on the nitrogen in the plant which is derived from the fertilizer, the tracer method is considered by most researchers to give a more accurate measure of fertilizer efficiency, where efficiency is defined in terms of amount of fertilizer-derived nitrogen in the plant. However, the most obvious limitation of this method is its expense and requirement for sophisticated analytical facilities. Less obvious limitations also may be inherent in the method. We refer to possible error in the interpretation of tracer data. Whether this error is significant is still open to question, and its discussion lies beyond the scope of this paper. For a more detailed analysis of tracer versus nontracer methods for measuring plant recovery of applied nitrogen, see Hauck and Bremner (1976) and Hauck (1978).

Linear regression methods have been applied to tracer and nontracer data in order to improve the statistical basis for the relationship between amount of nitrogen taken up to that applied. The method is discussed by Terman and Brown (1968); an example using tracer and nontracer data is given by Westerman and Kurtz (1974).

The direct tracer and indirect nontracer methods of determining plant recovery of applied nitrogen attempt to distinguish between the fertilizer- and soil-derived nitrogen in the plant. Another method of estimating fertilizer nitrogen recovery which does not distinguish between soil and fertilizer nitrogen uses the concept of "apparent recovery". The calculation is made by expressing the amount of nitrogen in the harvested portion of the crop as a percentage of the nitrogen fertilizer applied. Although this is a gross rather than net measure of nitrogen fertilizer efficiency, it provides a useful value when one is concerned mainly with measuring the total removal of nitrogen from the soil by the harvested portion of the crop.

Use of Efficiency Data

It is not our purpose here to select the concept of nitrogen fertilizer efficiency which we consider to be the most useful. However, we will illustrate how the several concepts can be used.

Table 6 presents calculations of nitrogen fertilizer efficiency based on three concepts of efficiency--percent removal by the crop of applied nitrogen, yield (dry matter) increase per increment of fertilizer nitrogen, and dollar return per dollar invested for fertilizer nitrogen. The calculations use information on yield of and nitrogen percentage in a coastal bermudagrass crop at 10 nonzero rates of nitrogen application. Yields of dry matter increased with increasing rates of nitrogen application up to about 1,350 kg nitrogen/ha. The percentage of total nitrogen in the plant increased with rate of nitrogen application, the percentage at the highest rate (2,016 kg nitrogen/ha) being twice that of the unfertilized crop. The nitrogen application rate of 224 kg/ha was the one most efficiently removed by the crop (65% of the total nitrogen applied) but did not produce the largest average yield increase per kilogram of nitrogen applied, nor the highest dollar return for increment of fertilizer dollar invested. An application of 112 kg nitrogen/ha gave the highest dry matter increase, hence dollar return per increment of nitrogen applied but not the highest profit. (Note that the economic analysis is based only on the market value of the crop and the cost of the fertilizer nitrogen based on May 1978 figures. The analysis does not consider the cost of fertilizer application or the slightly higher cost of harvesting the higher yields).

Percent recovery of the applied fertilizer nitrogen was calculated both on an incremental and cumulative basis. On an incremental basis, the percent recovery is expressed in terms of the amount of nitrogen in the plant that apparently was taken up for each increment of nitrogen applied. The yield and nitrogen concentration data in Table 6 indicate that increasing the nitrogen application rate from 1,344 to 1,568 kg nitrogen/ha had very little effect on the total nitrogen content of the harvested crop; less than 1% of the additional 224 kg of nitrogen was taken up. None of the additional nitrogen at the higher application rates apparently was recovered in the crop. On the contrary, less total fertilizer nitrogen was taken up because of adverse effects on plant growth.

Percent recovery on a cumulative basis is calculated in terms of the total fertilizer nitrogen in the harvested crop in relation to the total nitrogen applied. Notice the qualitative similarity of percent recovery values calculated in this manner as compared to the incremental method. However, quantitatively the values grow progressively dissimilar with increasing rate of nitrogen application. Most economists base their economic analysis of nitrogen fertilizer use on incremental values because this method of calculation makes obvious the dollar value of each fertilizer increment.

On the basis of percent plant recovery of applied nitrogen (incremental or cumulative), one could judge that maximum fertilizer efficiency was obtained at the 224 kg nitrogen/ha rate. From a plant physiological viewpoint, a greater percentage of the applied nitrogen was used in dry matter production at the 112 kg nitrogen/ha rate. The economist might regard the 1,120 kg/ha application rate to be of economic value, being concerned not with the highest return for dollar invested for nitrogen (incremental basis), but with the nitrogen application rate which gives maximum profit. Perhaps the economist would consider that an application of about 1,000 kg nitrogen/ha would provide a safer margin for experimental error, ensuring maximum profit. At application rates greater than 1,344 kg nitrogen/ha in this study, profits would have declined because yields were decreased and nitrogen was wasted. Engelstad (1963) concluded from an analysis of corn and cotton yields and profits that a rather wide range in nitrogen fertilizer rates is possible without seriously affecting either yield or profit. Those concerned mainly with the amount of fertilizer nitrogen left residual in the soil which is subject to leaching might judge the 224 kg nitrogen/ha rate to be the most efficient in terms of percentage removal of applied nitrogen, but might recommend the 112 kg nitrogen/ha rate because it left the least amount of nitrogen in the soil after harvest. This viewpoint would ignore the fact that the dollar return at the latter application rate would be less than one-fourth of that (about $162 versus $789) which would be obtained from the rate giving maximum profit.

In a second comparison of fertilizer efficiency concepts, we make use of data for small grains grown under irrigation presented in Table 7, which represents the results of multiple rate experiments at two locations in Montana and one in Arizona. At site 1, the 100 kg nitrogen/ha application rate could be judged most efficient from several viewpoints. This rate produced the highest grain yields; the crop contained more fertilizer nitrogen in the grain or whole plant and percent recoveries of applied nitrogen in the grain or whole plant were highest from this rate, regardless of whether they were calculated on an incremental or cumulative basis. However, in terms of grain increase per amount of nitrogen applied, or dollar return for fertilizer dollar invested, the 50 kg nitrogen/ha rate could be considered more efficient. From the viewpoint of profitability, one might choose the 100 kg nitrogen/ha rate. The next higher increment clearly was wasteful and inefficient, from any viewpoint.

The data from site 2 indicate that the highest application rate (150 kg nitrogen/ha) was the most efficient in terms of grain yield and nitrogen uptake in grain or whole plant. The data also strongly indicate that additional nitrogen would have produced higher grain yields and more profit, albeit at the 150 kg nitrogen/ha rate, about half of the applied nitrogen was not taken up. It is interesting to note that at both sites, half of the nitrogen applied at the rate considered to be most "efficient" remained in the soil or was not accounted for.

Data for site 3 are from a southwestern location with a higher yield potential for wheat than sites 1 and 2. Yet, the first increment of applied nitrogen (112 kg/ha) resulted in the highest recovery of nitrogen in the grain and the greatest economic return. The 224 kg nitrogen/ha application produced slightly more grain containing more nitrogen, but higher application rates up to 560 kg/ha caused

Table 7. Wheat grain yield and nitrogen fertilizer efficiency calculated on the basis of three concepts.

			Incremental				Accumulative	
Applied nitrogen	Yield	Nitrogen removed in grain	Removal of fertilizer nitrogen in grain	Grain increase	Economic return*	Total recovery grain & straw	Removal of fertilizer nitrogen in grain	Total recovery grain & straw
kg/ha	kg/ha	kg/ha	%	kg/kg N	$ output/ $ N cost	%	%	%
			Spring wheat, site 1**					
0	3,293	61	-	-	-	-	-	-
50	4,455	78	34	23.2	5.40	40	34	40
100	4,946	102	48	9.8	2.28	66	41	53
150	4,287	95	-	-13.2	-3.06	-	23	34
			Site 2**					
0	1,095	23	-	-	-	-	-	-
50	2,305	35	25	24.2	5.63	28	25	28
100	4,005	60	50	34.0	7.91	60	37	44
150	4,838	84	48	16.7	3.88	58	41	49
			Site 3***					
0	7,090	129	-	-	-	-	-	-
112	8,254	179	45	10.4	2.42	-	45	-
224	8,333	200	19	.7	.16	-	32	-
336	7,868	196	-	-	- .96	-	20	-
448	7,941	198	-	-	-	-	15	-
560	7,795	196	-	-	-	-	12	-

*Based on nitrogen at $0.43/kg and wheat at $0.10/kg.

**Calculated from data supplied by Dr. Neil Christensen, Montana State University, Bozeman, Montana.

***Calculated from data of Gardner and Jackson (1976).

yields to decline while the nitrogen content in the grain remained constant.

Comparing the data from sites 1, 2, and 3, one can note a marked difference in yields without added nitrogen which can be attributed to a large degree to differences in soil nitrogen supply. Even though the soil nitrogen level at site 3 was high, its greater yield potential compared to those of sites 1 and 2 permitted a marked response to fertilizer nitrogen such that at maximum yield, recovery of fertilizer in the grain approached 50% of the nitrogen applied. Thus, not only level of soil nitrogen, but all factors which determine yield potential determine the response of plants to applied nitrogen; that is, they determine the efficiency by which plants use fertilizer nitrogen.

Fertilizer Efficiency Versus Management Efficiency

The preceding discussion of the several concepts of fertilizer efficiency presupposes that by one method or another we measure the efficiency of applied nitrogen. When comparing different nitrogen fertilizers, one may be justified in comparing the relative efficiency of one fertilizer to that of another. But with a single nitrogen fertilizer, can one justifiably state that the fertilizer *per se* is more or less efficient, depending on how much fertilizer is applied, on how it is applied, or on other crop management factors? How can one state the efficiency of a single fertilizer in absolute terms when one knows that relative "efficiency" values are influenced by the crop and soil management system? With irrigated agriculture in particular, there is a close relationship between water use and plant use of applied nitrogen. All data relating to nitrogen fertilizer uptake and use should be evaluated on the basis of an intimate knowledge of the soil-fertilizer-crop management system. Therefore, it is not the efficiency of the fertilizer which is being measured, but the efficiency or effectiveness of the fertilizer management system.

The term, fertilizer efficiency, no doubt will continue to be used in many ways. It is our intention to draw attention to its ambiguous nature, and to suggest that users of this term clearly define the manner of its use. We agree with Terman (1976) that effective use of fertilizer nitrogen produces the "highest possible yields consistent with fertilizer supply, profit, and environmental effects".

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VOLATILE LOSSES OF NITROGEN FROM SOIL

D. E. Rolston[1]

ABSTRACT

Nitrogen may be lost from soil in the gaseous form by two major mechanisms, ammonia volatilization and denitrification. Ammonia gas may be lost to the atmosphere whenever ammonium compounds are applied to the soil surface. The greatest ammonia losses occur from calcareous soils at high soil pH. Fertilizers such as urea and ammonium sulfate result in greater ammonia loss than that from ammonium nitrate when applied to a moist soil surface. More than 50% of the applied fertilizer may be lost by ammonia volatilization if precautions are not taken. The best solution for minimizing ammonia loss is to incorporate or place ammonium compounds approximately 10 cm below relatively dry surface soil. The volatile products of denitrification, nitrous oxide and nitrogen gas, may be lost from the soil whenever the soil becomes wet enough that oxygen becomes depleted and sufficient carbon is available from organic materials to support microbial activity. Denitrification occurs significantly only over a very narrow soil-water content range near saturation and in those portions of the soil profile with fairly high organic material. Consequently, denitrification will generally only occur in the surface 60 cm of most soils of arid regions unless perched water tables exist at a buried surface horizon. Management practices to either minimize or maximize denitrification should be directed at controlling nitrate position and water content in the surface soil.

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INTRODUCTION

Nitrogen may be lost from soil in the gaseous form by two major mechanisms, ammonia volatilization and denitrification. Ammonia gas may volatilize to the atmosphere whenever ammonium containing materials are applied to soil. The volatile products of denitrification, primarily nitrous oxide and nitrogen gas, may be produced whenever the soil becomes so wet that oxygen becomes depleted. Ammonia volatilization is a chemical reaction whereas denitrification is a biological process of soil microorganisms. Both processes result in an economic loss to a grower.

Another mechanism which may result in volatile losses of nitrogen from soil is the production of nitrous oxide during the nitrification of ammonium compounds to nitrate (Bremner and Blackmer, 1977). This process has only recently been elucidated, and the magnitude of losses are not yet established.

AMMONIA VOLATILIZATION LOSS

Ammonia may volatilize to the atmosphere whenever ammonium or ammonium-forming inorganic or organic nitrogen materials are applied to soil. The ammonium may be derived from fertilizer or it may be derived from manures or organic wastes. Organic nitrogen must be mineralized to the ammonium form before ammonia volatilization can occur. For instance, manures placed on the soil surface would result in ammonium being mineralized from the organic form and be available for volatilization. Ammonium compounds can also reach the soil from rainwater or irrigation water. Besides an economic loss to a grower applying fertilizer or manure for crop production, the loss of ammonia to the atmosphere also results in a pollution problem inasmuch as the ammonia may eventually be absorbed by surface waters and contribute to the growth of algae in lakes and streams. Thus, it is desirable from both an agricultural and an environmental viewpoint, that ammonia volatilization losses be kept to the very minimum.

Measurement Methods

The primary method of measuring ammonia volatilization loss from soils is to place a cover or chamber over the soil surface, pass air through the chamber, and trap any ammonia gas released from the soil by bubbling the air stream through an acid solution (boric acid is commonly used). An experimental apparatus used extensively for research on ammonia volatilization loss from laboratory experiments is described by Fenn and Kissel (1973). Nearly all measurements of ammonia volatilization loss have been conducted in the laboratory where the soil is placed within an airtight chamber so that air from outside of the immediate soil cannot move into the system. The extrapolation of this procedure to the field is a more difficult task inasmuch as the process of using pressure or vacuum in moving air through the chamber over the soil surface may either underestimate or overestimate the actual ammonia loss, respectively.

An alternative approach (Denmead et al, 1974, 1976) for measuring the ammonia volatilization loss from soil surfaces is to measure the concentration of ammonia in the atmosphere above the soil or crop canopy. Concentration measurements in conjunction with meteorological measurements such as air temperature and wind speed at several heights above the soil are used to calculate the amount of ammonia gas escaping from the soil surface. The advantage of this method is that there is no disturbance to the soil environment and, thus, it gives an undisturbed value. The amount of loss would be representative of a much larger area than could be measured by placing covers over the soil surface. The ammonia volatilization loss of an entire field or basin is potentially within the realm of such a procedure.

Factors Affecting Loss

When ammonium compounds which have been applied either as fertilizer or have been mineralized from organic material such as manure or crop residues reach the surface of calcareous soil, it is likely that some of the compounds will react with calcium carbonate (lime) in the soil to form calcium precipitates. The product, ammonium carbonate, is unstable and subsequently decomposes to ammonia gas, water, and carbon dioxide. The ammonia gas given off may form ammonium hydroxide with the amount of ammonium hydroxide formed dependent upon the solubility of the calcium precipitate and the rate of formation. The formation of ammonium hydroxide in the vicinity of the ammonium fertilizer granules or solution results in an increase in the soil pH (becomes more basic) which tends to further increase ammonia gas evolution.

A complete description of the reactions resulting in ammonia gas production from ammonium compounds is very well described by Fenn and Kissel (1973). Fenn and Kissel (1973, 1974, 1976) have studied many of the factors affecting ammonia volatilization loss from soil. The two major factors affecting ammonia loss are soil pH and the source of the ammonium compound. The ammonia volatilization loss increases as the soil pH increases. It is expected that very little volatilization loss would occur from surface applications of ammonium compounds to acid soils with pH values less than 6. However, ammonia loss can occur in acid soils, as has been demonstrated by Du Plessis and Kroontje (1964). Considerable loss of ammonia can occur when ammonium compounds are applied to the surface of basic soil with pH above 8.

However, soil pH is not the only important factor in determining loss. Another important factor associated with absolute values of pH is the buffering capacity of the soil. The buffering capacity is the ability of the soil to withstand changes in pH when either acidic or alkaline compounds are added to the soil. The ability to withstand changes in pH is frequently associated with the amount of lime (calcium carbonate) in the soil. Sandy soils generally have a low buffering capacity, whereas loam or clay soils have considerable buffering capacity. Another important factor associated with pH is not the initial pH or the buffering capacity of the soil, but the upward pH shifts which occur near the band or granule of fertilizer resulting in increasing ammonia volatilization loss. Various ammonium compounds react differently in soil in terms of the pH shift near the fertilizer particle. Figure 1 shows the relative ammonia volatilization loss at two times after application from three ammonium compounds used as fertilizers. The compounds for which the calcium reaction products have low solubilities result in large ammonia losses whereas the calcium compounds formed that have relatively high solubilities result in low amounts of ammonia volatilization. This is well demonstrated in Figure 1 for two common fertilizer materials, ammonium sulfate and ammonium nitrate. Calcium sulfate, which is the reaction product from ammonium sulfate, has a very low solubility and precipitates, thus driving the reaction producing ammonium hydroxide and eventually ammonia gas. However, calcium nitrate which is the reaction product of ammonium nitrate is very soluble and decreases the reaction to ammonia resulting in low ammonia losses. Thus, the solubility of the calcium reaction products has a strong effect on the pH shift near the fertilizer material and greatly affects the volatilization loss. Ammonia volatilization from urea is very much dependent upon the soil surface conditions at time of application since urea will not convert to ammonium in a relatively dry soil. If urea is placed on the surface of a wet soil, however, conversion to ammonium will occur with a subsequent rapid ammonia volatilization. Ammonia volatilization from urea is directly related to initial soil moisture content due to the effect of water content on the duration of drying and the time for hydrolysis (Ernst and Massey, 1960). Figure 2 demonstrates the influence of water content on ammonia loss from surface-applied urea. There is generally some time lag in

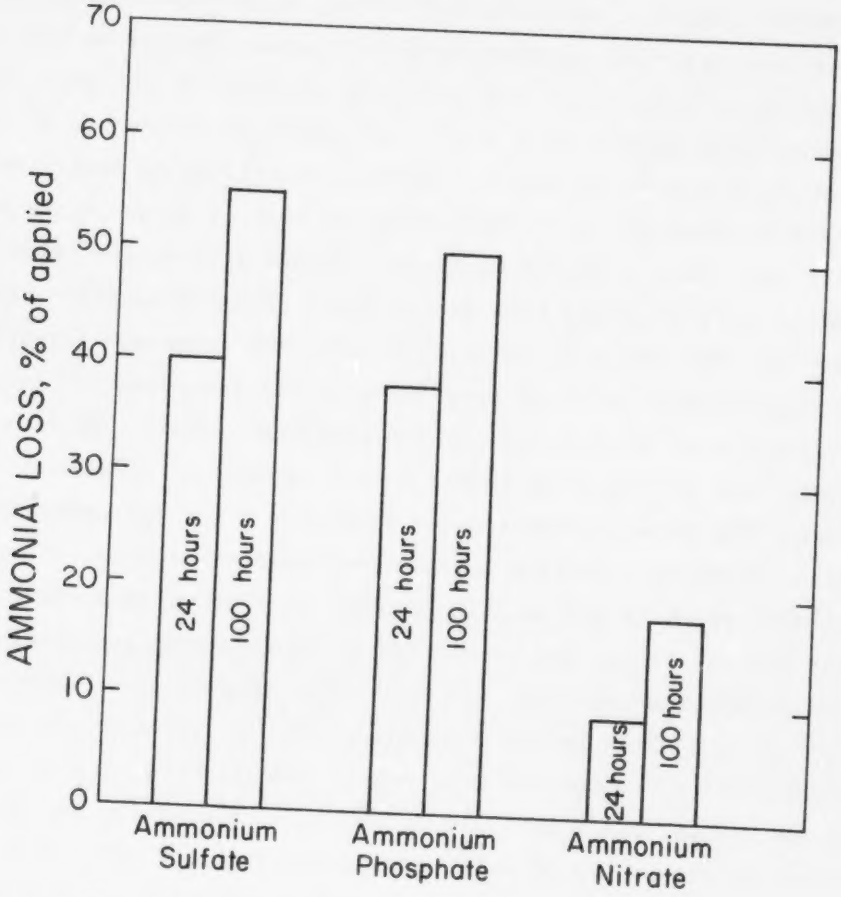

Figure 1. Total ammonia loss at the end of 24 and 100 hours at 22C for three ammonium sources. Ammonium was applied on the soil surface at 550 kg nitrogen/ha. (Redrawn from Fenn and Kissel, 1973).

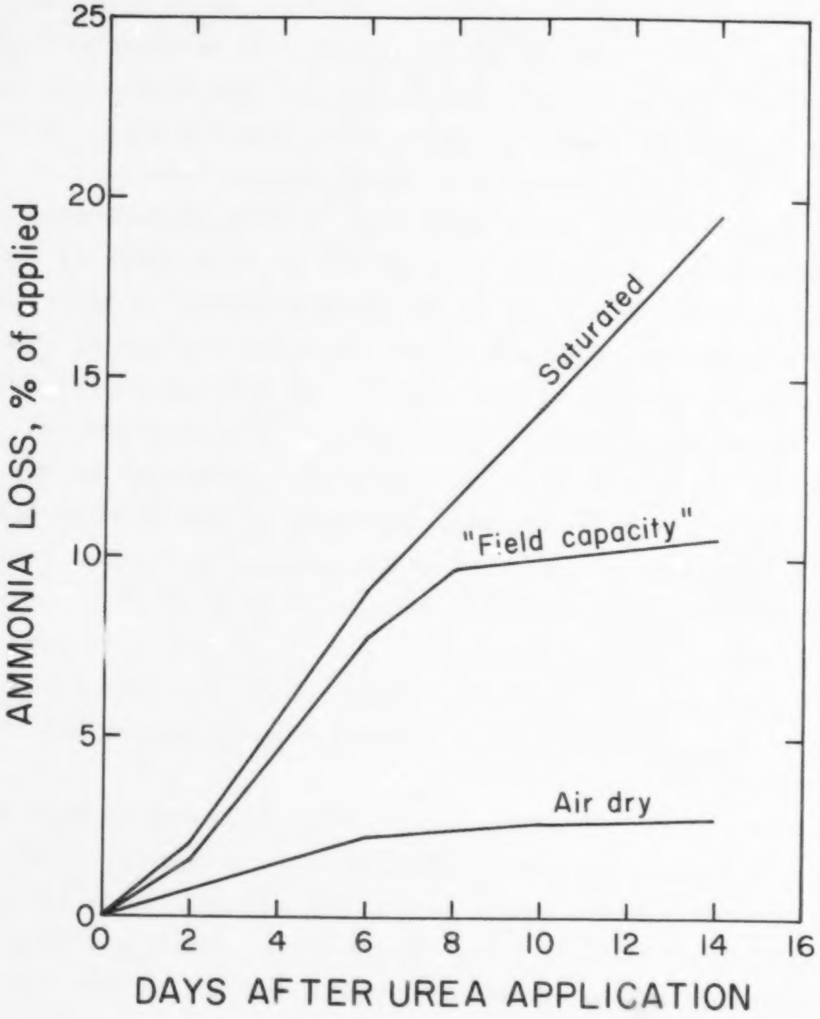

Figure 2. Total ammonia loss of nitrogen added as urea (109 kg nitrogen/ha) to a silt loam soil (pH 6.5, 24C) of differing initial water content as a function of time after urea application. (Redrawn from Ernst and Massey, 1960)

the hydrolysis of urea to ammonium. However, after hydrolysis occurs, urea has a greater ammonia volatilization loss in moist soil than that of the other compounds inasmuch as the calcium reaction product, calcium carbonate (lime), is less soluble than the other calcium reaction products and results in large ammonia losses.

Ammonia volatilization loss tends to increase as temperature of the soil surface increases, although not as drastically as that which occurs with changes in soil pH or ammonium source. A major point in terms of temperature effects on ammonia volatilization is that considerable volatilization loss may occur at temperatures as low as 12C.

The rate that ammonium fertilizer is applied to the soil has an effect on the ammonia loss for some of the compounds, but has very little or no effect on the loss from other compounds (Fenn and Kissel, 1974). For instance, an increase in ammonium application rate from approximately 50 to 500 kg of nitrogen/ha resulted in an increase of from 20% loss to approximately 45% loss in 100 hours for ammonium sulfate and ammonium phosphate. However, the percentage loss of ammonia from ammonium nitrate remained at approximately 20% regardless of application rate.

The time that the material is on the soil surface also has a slight effect on the ammonia volatilization (Figure 1). The greatest loss, however, occurs within 24 hours after application of the ammonium compound to the soil surface. Of the common fertilizer materials, ammonium sulfate and ammonium phosphate resulted in very little increased ammonia loss after 24 hours. However, for ammonium nitrate, the ammonia loss approximately doubled from 24 to 100 hours after application to the soil surface. As discussed before, the exception to this very rapid loss may be the loss associated with application of urea to soils in which there is a delay in the ammonia volatilization due to the necessity of converting urea to ammonium. This delay in volatilization from urea may be approximately two days.

Since considerable ammonia loss can occur when ammonium compounds are applied to the surface of soils, another important factor affecting volatilization loss is the depth of placement of ammonium compounds below the soil surface or the depth of incorporation into the soil. Smaller losses of ammonia occur if the ammonium compounds are placed below the soil surface than if the compounds are simply mixed uniformly in the surface soil as would occur as a result of harrowing or discing. This is due to the fact that some ammonium remains close to the surface and may result in ammonia loss. The moisture content and texture of the soil over the fertilizer material has an effect on ammonia volatilization. Dry soil over the fertilizer material is generally more effective in reducing ammonia losses than is a very moist soil above the zone of placement. This is primarily due to the fact that water, moving to the soils surface by evaporation, also carries along ammonium and results in ammonia volatilization, whereas very little or no movement of water to the soil surface occurs through dry soil. A clay or loam soil over the ammonium compound decreases the ammonia loss much more than does that of sand. This is due to the greater capacity of a clay soil for adsorbing ammonium than that of a sand. The ideal depth which generally insures that very little or no ammonia loss will occur if fairly dry soil is placed over the ammonium compounds is approximately 7- to 10-cm. Thus, placement below a relatively dry soil surface is the primary management factor for the minimal ammonia loss.

Rates and Total Amounts of Ammonia Loss

Rates and total ammonia volatilization losses vary greatly depending primarily upon the source of the ammonium compound, the initial pH of the soil, and the placement or incorporation of the material below the soil surface. Other factors causing lesser effects are the time after fertilizer application, the temperature, and the rate of fertilizer application. Diagrams showing the general effects of these factors on ammonia volatilization are given by Figure 3. Figure 3

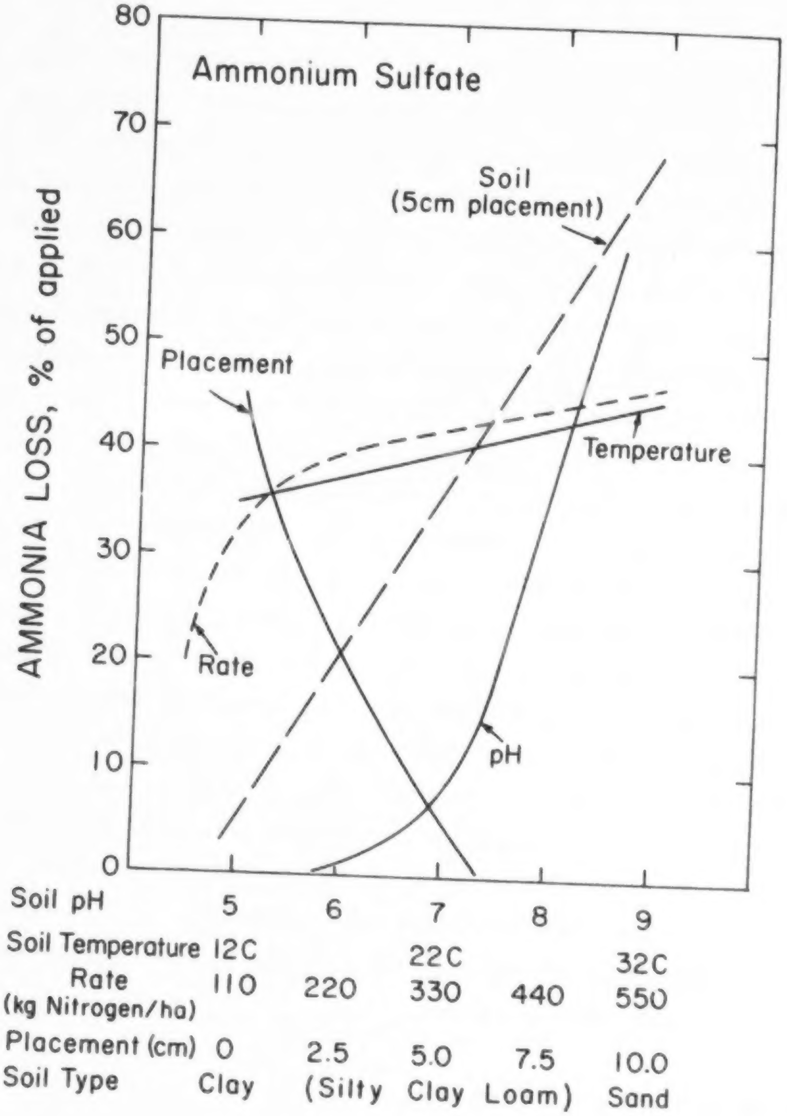

Figure 3. Approximate relationships of ammonia loss to soil pH, soil temperature, ammonium application rate, depth of placement, and soil type for ammonium sulfate. The relationships are for ammonia loss from a clay soil (excluding soil type curve), surface applications (excluding placement curve, 550 kg nitrogen/ha ammonium application rate (excluding ammonium rate curve), 30C (excluding temperature curve), and pH 7.6 (excluding pH curve).

shows the effects of soil pH, placement, soil types, temperature, and rate of application on ammonia loss from ammonium sulfate, a common material resulting in large losses. Ammonium nitrate gives smaller absolute amounts of loss than does ammonium sulfate and is affected by time and rate of application in a somewhat different manner. For example, percentage loss of ammonia from ammonium nitrate does not change as the rate of ammonium nitrate application increases. Another exception for ammonium nitrate is that substantial loss continues after 24 hours as compared to only small loss after 24 hours for ammonium sulfate. An important fact is that the absolute amount of ammonia loss can vary from 0 to 70% of the total ammonium fertilizer applied.

Practices to Prevent Loss

The major practice to prevent loss of ammonia by volatilization from ammonium compounds is to place or incorporate the materials approximately 10 cm below the soil surface. Placement of the fertilizer 10 cm below the soil surface is more effective in preventing loss than is incorporation uniformly within the top 10 cm of soil. Incorporation in the soil should take place immediately after broadcasting or placing the material on the soil surface, inasmuch as the maximum rates of ammonia loss occur very quickly after application. If the soil surface is very dry and the fertilizer materials are applied dry, the material can be left on the soil surface for a few hours until incorporation can be accomplished. However, a rain or nightly dew could solubilize the materials and begin ammonia volatilization. In general, ammonium compounds should not be applied to the surface of a wet soil, since ammonia volatilization loss begins immediately after application. Even if the material is placed below the soil surface, a wet surface soil may cause some loss by movement of ammonia to the surface with the evaporating water.

If circumstances prohibit placement below the soil surface or incorporation of the ammonium compounds in the soil, other practices can be used to minimize the ammonia volatilization loss. Using ammonium nitrate, for example, on the soil surface, would result in much less volatilization loss than if urea, ammonium sulfate, or ammonium phosphate were broadcast on the soil surface. Another means of minimizing loss for those circumstances where the material cannot be placed or incorporated, is applying the material to the soil surface and immediately irrigating in order to move the material slightly below the soil surface. This approach is not an entirely effective means of minimizing loss, inasmuch as the ammonium compounds will adsorb on the soil and move only a very small distance into the soil profile (a matter of one or two centimeters). The exception to this is urea. Urea must be hydrolized to ammonium before the strong adsorption processes will occur. Thus, if urea is applied to the soil surface and the soil immediately irrigated, urea would move far enough into the soil profile to minimize the loss.

A relatively common practice for applying nitrogen in irrigated agriculture is to apply the nitrogen materials along with the irrigation water. This practice may result in considerable ammonia volatilization if the nitrogen is applied as ammonium. Ammonia loss is greatest if ammonium compounds are applied through a sprinkler irrigation system due to evaporation than that which occurs from furrow and drip irrigation systems. Unpublished data of Rolston indicate that very little loss of ammonia occurred when ammonium sulfate was applied through a drip irrigation system, due primarily to the fact that the ammonium was applied at a point and most of the material moved below the soil surface. Ammonium nitrate is less susceptible to loss than is ammonium sulfate or ammonium phosphate and would result in smaller losses if applied with irrigation water. Urea is the most desirable compound to apply through irrigation systems, inasmuch as it must be hydrolized to ammonium before ammonia can be volatilized.

DENITRIFICATION LOSS

Denitrification is the biological reduction of nitrate and nitrite to several volatile gases. These gases may be nitric oxide, nitrogen dioxide, nitrous oxide, and molecular nitrogen. However, in most field situations where oxygen is present at least in some portion of the soil profile, the two gases released are nitrous oxide and molecular nitrogen. Denitrification is accomplished by bacteria capable of using nitrate in place of oxygen. Thus, whenever the soil becomes wet enough that oxygen is excluded from part of the soil profile, the potential for denitrification exists. Other necessary conditions for denitrification are the presence of nitrate and soluble carbon in the zone which has been depleted of oxygen. Carbon is derived from soil organic matter, manures, or plant residues. The gases released from denitrification are harmless to human health at the concentrations that would be emitted into the atmosphere. However, nitrous oxide gas may contribute to the destruction of the ozone layer of the lower stratosphere. The destruction or decrease in the amount of ozone in the stratosphere would result in increased incidents of skin cancer and other biological effects due to increased ultraviolet radiation reaching the earth. It has been postulated that increased use of nitrogen fertilizers will contribute to the depletion of the ozone layer. However, that has not been substantiated inasmuch as the magnitude of the total earth's sources and sinks for nitrous oxide has not been elucidated. Denitrification would constitute an economic loss to a grower fertilizing with either commercial fertilizers or manure but would be advantageous for decreasing possible nitrate pollution to the groundwater from heavy applications of sewage effluents, sewage sludge, or manure to soil.

Measurement Methods

The amount of denitrification is generally the unknown in attempts to evaluate the fate of nitrogen fertilizers or wastes applied to soils. Denitrification is usually calculated by difference from measurements of the other components of the nitrogen cycle, such as fertilizer addition, plant uptake, leaching, and residual soil nitrogen. Determination of denitrification by difference is commonly used inasmuch as the various components of the nitrogen cycle, such as fertilizer addition and plant uptake, have been extensively and relatively easily measured. The techniques for measurement of these various quantities is relatively well known and familiar to people. However, determining denitrification by difference is at best no better than the reliability of the other measurements with all errors accumulating in the difference value. The measurement of such components as the residual soil nitrogen remaining in the soil after fertilization and the amount of nitrogen which has leached from the soil profile is very difficult to accurately measure, especially over relatively large land areas. Rolston and Broadbent (1977) and Biggar and Nielsen (1976) have shown that the leaching component, even from very small field plots, is very difficult to measure accurately. In addition, determining denitrification by difference generally does not allow evaluation of the rates or the dynamic nature of the denitrification process. It also does not allow for determination of that proportion of denitrification consisting of nitrous oxide, a potential environmental concern.

Field methods for measuring denitrification other than by difference have required placing a sealed compartment over the soil surface and either trapping or sampling the gases evolved. This method can measure nitrous oxide evolution reasonably well. However, it is difficult to determine how much nitrogen gas has evolved because small increases above the ambient atmospheric concentration of 78% nitrogen cannot be measured. Direct methods of measuring the volatile gases from denitrification have been used by Rolston et al (1976) and Rolston and Broadbent (1977) by using large quantities of isotopically tagged nitrogen in very small field plots. The use of the stable isotope, ^{15}N, to tag the fertilizer nitrogen allows the nitrogen gas to be measured in the soil atmosphere and within compartments placed over the soil surface. Since nitrous oxide has a low concentration (0.3 ppm) in the atmosphere, it can be easily measured within the soil and in a chamber over the soil. Measuring denitrification directly has the disadvantages of a very expensive means of determining denitrification and, thus, is limited to relatively small field plots or laboratory experiments. Another disadvantage of the direct method using tagged nitrogen is that the method requires fairly complicated measurement techniques, experienced technicians, and special instrumentation. The advantages of the direct method are that both nitrous oxide and molecular nitrogen from denitrification can be measured as well as total denitrification. Thus, the time period over which denitrification occurs can be elucidated in order to develop management decisions to control denitrification.

Factors Affecting Loss

As with all biological and physical processes in the field, several soil parameters affect denitrification. The most important factors are the water content of the soil, the development of pockets or zones where oxygen is excluded, the amount and availability of carbon, soil temperature, and soil pH. Each of these factors will be considered in turn.

Water content influences denitrification by providing a suitable environment for microbes and through blockage of the soil pores reducing the ability of soil to transmit oxygen to the zones of high microbial activity and respiring plant roots. As the soil pores become nearly filled with water, oxygen can reach the sites of high oxygen consumption only by diffusing through the water, which is a very slow process, or by diffusing through a few large pores. The greatest potential for denitrification would be for completely water saturated soil. In the field, soil is saturated for only short periods during an irrigation or rainfall, or the soil may become saturated directly above clay pans or impeding layers in the soil profile. In many irrigated soils of semiarid regions, only the upper few centimeters (0-10-cm depth) are truly saturated during an irrigation or rainfall. All depths below the surface few centimeters will be slightly unsaturated with some pores filled with air through which oxygen may diffuse rather rapidly. Denitrification occurs over a very narrow range of soil-water contents or soil-water tensions. No denitrification apparently occurs when soils become drier than field capacity (approximately 30 centibars tension) as discussed by Focht and Verstraete (1977). Rolston et al (1976) and Rolston and Broadbent (1977) have shown for an alluvial loam soil that very little denitrification occurred at soil-water tensions greater than 10 centibars (field capacity is generally considered to be approximately 30 centibars and the wilting point is 1500 centibars). For a clay soil, the upper limit of tension for which denitrification may occur would be expected to be approximately 15 centibars, whereas in a sandy soil the upper limit may be approximately 5 centibars.

The development of soil zones with all oxygen excluded is another important factor affecting denitrification loss. Not only is the total volume of soil without oxygen important, but also the position of that zone within the soil profile. In deep uniform soils with a water table greater than 10 or 20 meters, it would be expected that the zone of lowest oxygen levels would occur near the soil surface. This has been shown by Rolston et al (1976) and Rolston and Broadbent (1977) for a deep loam soil with the water table at approximately 25 meters. Oxygen levels were smallest within the top 15-30 cm of soil than anywhere else in the soil profile. This is due primarily to the fact that the upper portion of the soil profile becomes wettest due to generally higher bulk densities and an impermeable layer for water infiltration at the surface and to high oxygen use by microbes and roots in the surface. Other positions which may become excluded of oxygen are above slowly permeable layers, such as hard pans, or clay pans within the soil profile. In many soils, however, it may be unlikely that soils above clay pans become anaerobic due to insufficient carbon or microbial activity at the depth of the perched water table.

Denitrification will not occur unless sufficient available carbon is present regardless of the amount of nitrate present or lack of oxygen in the soil. Thus, the amount of carbon, its position in the profile, and its position in relation to the nitrate of the soil, will have a large influence on the rates and amounts of denitrification. Percentages of carbon are highest at the soil surface and decrease rather rapidly with depth in most soils of semiarid regions. Exceptions are peat or high organic matter soils, such as the Sacramento-San Joaquin Delta of California. Thus, as nitrate is moved by irrigation water or rainwater to deeper depths within the soil profile the chances of any substantial denitrification decrease. Rolston (1976) measured no denitrification in a Yolo loam soil kept near saturation at the soil surface after the nitrate had leached from the upper 60-cm of soil. Pratt et al (1976) determined that denitrification was limited mostly to the top 60-cm of three soil profiles (sandy loam, sandy loam with clay pan at 60-cm, and silty clay soils) to which manure had been applied for four years. An exception to the greatest denitrification occurring in the upper few cm of soil would be those cases where a buried profile containing considerable organic matter lies somewhere within the soil profile and at a position such that a perched water table would occur.

The increase in denitrification from the addition of carbon is well established (Bremner and Shaw, 1958). The effect of plant roots in accelerating denitrification is also well known (Stefanson, 1972; Volz, 1976; Woldendorp, 1962). Rolston and Broadbent (1977) also found that very little denitrification occurred in a loam soil maintained nearly water saturated and uncropped. The presence of a crop not only contributes organic material from dead roots but also decreases oxygen concentrations through root respiration. Thus, the presence of a crop growing on the soil at the time fertilizer is applied can be expected to increase the potential for denitrification. The length of time that plants have been growing on the soil also greatly increases denitrification. Rolston and Broadbent (1977) showed much smaller rates of denitrification from a plot to which grass had been planted approximately three months prior to the experiment than did those from experiments (Rolston et al, 1976) where grass had been grown in the plot for approximately two years. With grass on the plot for a long time period, the amount of organic material added to the soil from the root system was apparently large. The addition of other organic sources to the soil will also increase the potential for denitrification. For instance, incorporating manure into the soil would increase carbon levels and potentially could result in increased denitrification. However, the manure would most likely have to be left in the soil long enough for sufficient carbon to become available to induce denitrification. As discussed previously, the nitrate must be at the same position in the profile as the carbon in order for denitrification to occur. Denitrification increases as the soil temperature increases due to increased biological activity. Rolston and Broadbent (1977) determined that denitrification was still measureable in plots to which manure had been added at a soil temperature of 8-10C. Thus, denitrification may occur in the winter months if other conditions are optimum.

Soil pH influences the rate of denitrification and the proportion of the different gases released (Focht and Verstraete, 1977). Denitrification is small at very low pH and increases as pH increases (Broadbent and Clark, 1965). However, in the pH range between 6 and 8, total denitrification is only slightly affected by changes in soil pH.

Rates and Total Amounts

The rates and total amounts of denitrification are dependent upon each of the factors described in the previous section. The total denitrification occurring from a nitrate application of 300 kg nitrogen/ha to a Yolo loam field soil as affected by carbon level, water content, and temperature are given by Figure 4 (Rolston and Broadbent, 1977). The rates measured by Rolston and Broadbent (1977) vary from 0 to 70 kg of nitrogen/ha/day. The denitrification rate of 70 kg nitrogen/ha was for a plot maintained nearly saturated and into which 34 metric tons of manure/ha had been incorporated in the upper 10 cm of soil approximately three weeks before the nitrate fertilizer was applied to the wet soil surface. The total amount of denitrification for this manure treatment, representing nearly maximal denitrification potential, was approximately 200 kg of nitrogen out of 300 kg applied. Most of the denitrification occurred in the first five days after the fertilizer was applied. The soil was kept continuously wet (near saturation) during this time period. No denitrification could be detected after approximately twenty days. This time period corresponds approximately to the time that the nitrate would have been leached from the top 30-45 cm of soil. The maximum rate measured in a cropped field plot (perennial rye grass) by Rolston et al (1976) was approximately 10 kg of nitrogen/ha/day. Again, most of the denitrification occurred over a 30-day period with approximately 150 kg of nitrogen denitrified out of a total of 300 kg applied. For uncropped plots with no carbon addition and maintained close to saturation, the maximum denitrification rate measured was approximately 2.5 kg of nitrogen/ha/day with the total loss occurring over a 15-day period of only 8 kg of nitrogen out of a total of 300 kg applied. Maintaining the soil constantly wet without allowing for redistribution, evaporation, or plant uptake represents the optimum conditions for denitrification in terms of water content and anaerobic development. The denitrification occurring during and after normal irrigation or

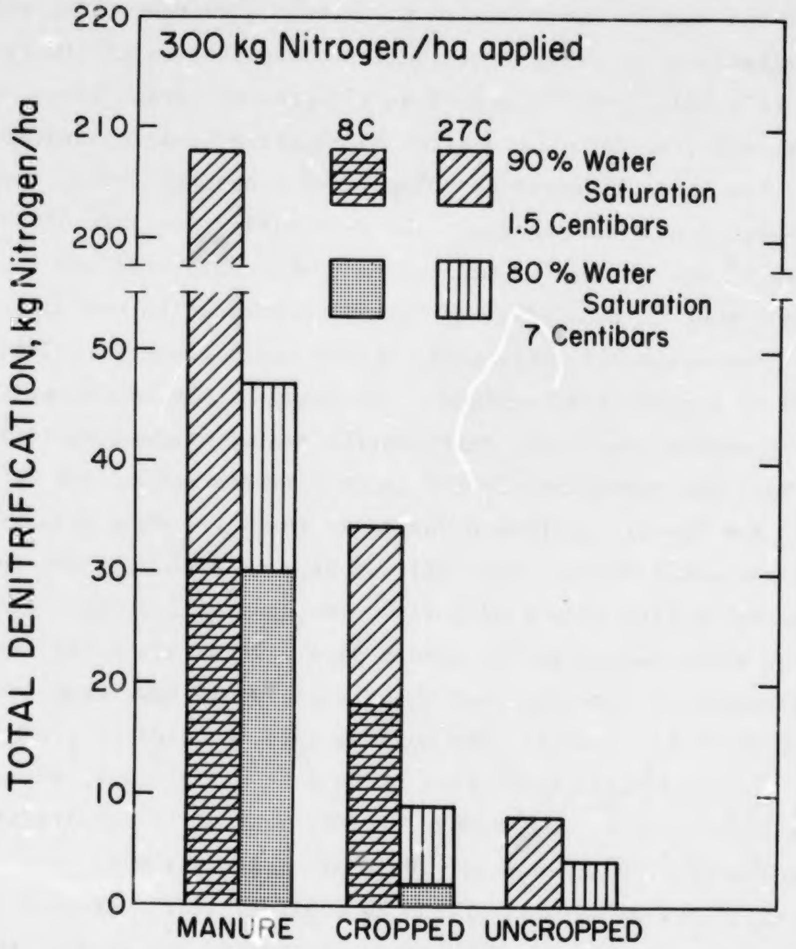

Figure 4. Total denitrification from cropped, uncropped, and manured field plots (Yolo loam) at two soil temperatures and two water contents.

rainfall events should be considerably smaller than these values inasmuch as the soil profile would be less wet and less anaerobic over shorter time periods. An estimate of the expected denitrification that might occur over an entire season may be calculated from the rate experiments of Rolston and Broadbent (1977). The assumptions used in the calculation are as follows: 1) The denitrification flux during wet periods is 5 kg of nitrogen/ha/day as measured for the nearly saturated, cropped plots of Rolston and Broadbent (1977); 2) The above flux occurs for 12 hours during an irrigation cycle or heavy rainfall event; 3) The above flux occurs for the first four irrigations of a season and for one rainfall event. From these assumptions, denitrification was calculated to be approximately 13 kg of nitrogen/ha. If 150 kg of nitrogen/ha was applied as fertilizer, denitrification loss would be about 8%. Although, little evidence exists for the above assumptions, the percent denitrification is the same order of magnitude as values determined from nitrogen balance experiments. Broadbent and Carlton (1976) found that denitrification from a three-year nitrogen balance field study was between 5 and 8% per year on a sandy loam soil cropped with maize.

Management Practices

Management of fertilizer nitrogen or nitrogen from wastes applied to soils may be directed in either of two directions. One goal would be to minimize denitrification losses, such as would be the objective of a grower applying fertilizer and wanting to maximize the use of that fertilizer. The other management objective might be to increase denitrification to the maximum for applications of large amounts of sewage effluent, sludge, or manure to soil in order to minimize the amount of nitrate leaching below disposal sites. In order to minimize denitrification, the major factors affecting denitrification should not occur at the same point in the soil or at the same time. For example, high concentrations of nitrate should not be allowed to occur at a position in the profile with high concentrations of carbon and high water contents. The absence of either nitrate or carbon or the presence of oxygen would result in very little or no denitrification. Thus, management practices should be directed at managing the water, the carbon, and the fertilizer such that they do not occur simultaneously in the soil profile. A proper management decision would be that fertilizer should not be applied to the surface soil (high carbon) at times when water contents are likely to be very close to saturation. Thus, one would not want to apply fertilizer to the surface of soils in winter or fall due to the likely chance that water contents would become large along with high amounts of carbon in the surface portion of the soil profile. Another management decision to decrease or eliminate denitrification would be to insure that fertilizer was placed at such a depth where plant roots could still obtain the needed nitrogen yet not be in a zone of high carbon or potentially high water contents. Application of manure along with nitrogen fertilizer would be an unrealistic practice if one wanted to minimize denitrification. If water contents or tensions could not be maintained outside the range where denitrification occurs, it may be possible to time fertilizer application so that nitrate will not occur in the zone containing high carbon at a time when water contents would be close to saturation. For instance, it may be possible to apply ammonium fertilizer immediately before a large irrigation for wetting the profile at the beginning of the season. The next irrigation would be timed and managed such that the nitrate would be moved out of the zone where most of the carbon occurs yet remain within the zone where roots could obtain the nitrogen.

Opposite kinds of management decisions would be made to maximize denitrification. One would want to have high water contents and thus low oxygen and high carbon levels in the zone containing nitrate. This could be accomplished by keeping the soil very wet by water management and adding carbon materials, such as manure, sludge or crop residues to the same zone where the high water contents and high nitrate concentrations would occur. One would also have to consider that for disposal of wastes, the nitrogen is generally in the ammonium form. Thus, one would have to allow a cyclic period of aerobic conditions for ammonium to transform to nitrate. Such a system to maximize denitrification becomes fairly complex when the nitrogen is applied as ammonium.

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LEACHING OF NITRATE FROM SOILS

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ABSTRACT

Nitrate leaching from irrigated croplands reflects the superimposing of water management on the nitrification, crop removal, and denitrification processes. Rapid movement of the nitrate anion through most irrigated soils results from its repulsion or "negative adsorption" by negatively-charged soil particles. Under selected conditions (e.g., in high-clay soils), nitrate may appear to move even more rapidly than does the water in which it was present initially. It is only in a relatively few highly-acidic or highly-weathered soils that nitrate may actually be adsorbed by positively-charged sites.

Calculation of soil solution nitrate concentrations, and of mass emissions of nitrate, are illustrated. Estimation of average water flux from nitrate/chloride ratios, and from drain field outflows, are discussed. The high potential for nitrate leaching from irrigated animal and municipal waste disposal sites is mentioned, with stress being placed on the high denitrification potential at many such sites, because of high soluble carbon loadings, high soil microbe levels, and rapid oxygen depletion.

Root-zone and subsoil nitrate-nitrogen values are provided for typical croplands of southern California and central Washington.

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Such values are related to irrigation management and to downward water flux. Crop differences in ability to utilize all soluble nitrate from the soil solution are stressed, and some management alternatives for preventing deep percolation losses of nitrate are detailed.

Predictions of nitrate leaching losses from data for nitrogen application rate and drainage water volume are discussed. Typical predictions are presented and are compared to experimentally-measured values. Overprediction of nitrate leaching estimates for some furrow-irrigated tracts in the Pacific Northwest is demonstrated.

INTRODUCTION

Nitrate leaching represents the superimposing of water management on the nitrification, crop removal, and denitrification processes discussed in other chapters of this volume. Except in cases where fertilizer nitrogen has already been provided in the nitrate form, production of nitrate represents the end step during the transformations of nitrogen in well-aerated soils, whether the original nitrogen came from inorganic fertilizers, from crop residues or organic wastes, or from soil organic matter. It must be recognized from the outset that substantial soil solution nitrate concentrations are the rule, rather than the exception, during the growing season for well-managed, "fertile" soils. Soil nitrate, once produced, can be removed from the crop root zone either by plant uptake, by gaseous denitrification losses from poorly-aerated soil zones, or by nitrate leaching whenever excessive irrigation water is applied.

NITRATE MOVEMENT

The vast majority of irrigated soils are negatively charged.

Hence, the nitrate anion is repelled from soil colloid surfaces, and can be readily leached by percolating waters. The process whereby anions are repelled by charged particle surfaces is termed anion repulsion or "negative adsorption". Such repulsion is greater for more highly-charged soils (e.g. those having higher cation-exchange capacities), and usually increases with increasing soil pH. A few anions, such as phosphate and arsenate, move only slowly through soils, for they are either rapidly adsorbed on soil particle surfaces or are readily precipitated as slightly-soluble compounds. Anions such as nitrate and chloride, however, tend to be repelled by all but the least highly-charged or most acidic (e.g., most highly-weathered) soils.

In cases where highly-weathered or highly-acidic soils are being irrigated, one should be aware that nitrate may move less readily. Such soils would generally have pH values less than 5.5 to 6 (Espinoza et al, 1975), as well as the dark brown or deep red colors indicative of highly-weathered soils. Nitrate adsorption is also high in typical soils having high contents of amorphous minerals of volcanic origin (Kinjo and Pratt, 1971).

That nitrate will normally be repelled from particle surfaces, however, has been demonstrated by numerous studies. For example, Smith and Davies (1974) found only one Alabama subsoil (of pH 5.1) from among a group of 16 U.S. surface soils and subsoils that exhibited anion retention, whereas the other 15 samples (of pH 6.2 to 8.7) all evidenced anion repulsion to varying degrees. Because of this repulsion, they found that nitrate and bromide anions moved 1.1 to 1.6 times more rapidly through the samples than did the water in which the anions were present initially.

Two explanations can be offered for such enhanced mobility of anions. Because of anion repulsion, a lower concentration of nitrate exists in the water immediately adjacent to charged soil surfaces, and the anion must move through only a portion of the water in each pore sequence as it traverses the pore. It is also in the central part of the pore that the pore water velocity is greatest, so each anion is moving, on the average, more rapidly than is an average water molecule during bulk flow of water and solutes through a soil.

Because of anion repulsion, experiments in which a given amount of nitrate or chloride is introduced to a column of soil simultaneously with a given amount of "tagged" water commonly result in the nitrate or chloride anions appearing in the leachate before the corresponding increment of water appears. Such behavior is demonstrated in Figure 1. In either case, whether the anion was introduced as a single pulse of material or as a continuous source, the anion emerged from the soil column more rapidly than did the "tagged" water in which the anion was present initially. Such behavior is of the greatest concern in highly-charged, high-clay soils, such as the "black clays' of southeastern Texas (Thomas and Swoboda, 1970). Water movement through such soils is generally through fine pores, where the influence of the charged soil surfaces is particularly evident. The effect of anion repulsion on relative movement of water and nitrate through moderate- to coarse-textured soils is often negligible, for the nitrate and the water move essentially in tandem through relatively large water-filled pores. The key factor is the extent to which the pore transmitting most of the water are large relative to the thickness of the layer from which anions have been repelled near soil particle surfaces.

Anions such as chloride or bromide are used commonly as "tracers" of nitrate movement through soils, because they enter into relatively few biological reactions. Actual movement of "pulses" of nitrate through soils may be retarded, more diffuse, and of lower peak concentration than for these more inert analogues, however. This is due to the highly reactive nature of nitrate in actual biological systems, such as soils (e.g., Smith and Davies, 1974).

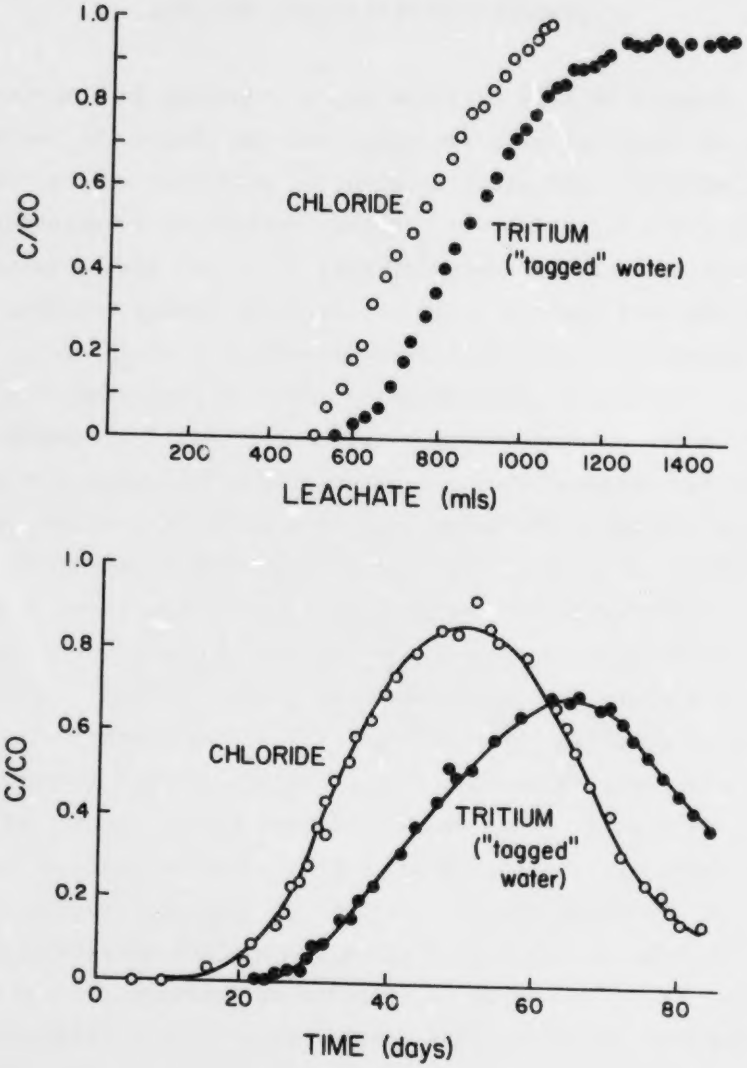

Figure 1. a. Top. Chloride and tritium ("tagged" water) breakthrough curves for a distrubed core of Maury soil (from Thomas and Swoboda, 1970).

b. Bottom. Relative chloride and tritium ("tagged" water) concentrations measured at a depth of 46 cm in some field plots (from van de Pol et al, 1977).

NITRATE CONCENTRATIONS AND FLUX

Results of soil analyses may be reported in a variety of ways. Some of these may be misleading, and may create the impression that a relatively minor nitrate pollution potential exists, when in reality the potential for nitrate leaching is substantial. It must be kept in mind, for example, that it is the soil solution, rather than the soil itself, which is displaced during leaching. Hence, information on residual nitrate levels in a soil before, during and following the cropping season must be converted to a solution basis prior to comparisons between studies. For example, a typical value for residual soil nitrate-nitrogen following a crop growing season may be on the order of 3 to 6 parts per million parts of dry soil. As a 30-cm depth of "average" soil weighs roughly four million kilograms per hectare, this means that 10 to 25 kilograms of soluble nitrate-nitrogen are present in each 30-cm depth of soil and might be leached whenever excess irrigation water or rain-water is supplied. Furthermore, the average water content of field soils is commonly on the order of only 15 to 30% (expressed on a dry-soil basis). This means that the nitrate concentration of the soil solution is 3 to 7 times higher than the level of residual soil nitrate (assuming that the latter was also expressed on a dry-soil basis). Hence, our 3 to 6 parts per million nitrate-nitrogen in the dry soil becomes 10 to 40 parts per million nitrate-nitrogen in the soil solution. Any displacement of this solution deeper into the soil profile can lead to appreciable environmental hazards.

Substantial concentrations of nitrate-nitrogen persist in the plant root zone of most "fertile" soils. Such concentrations need not constitute an environmental hazard, however, if no soil solution is displaced from the root zone. It is only the necessity of providing excess water for salinity control, and the inefficiency of water application in the field, which leads to the nitrate leaching associated with many irrigated croplands.

The amount of excess water passing through the crop root zone varies substantially from one area and from one type of irrigation system to another. For well-managed sprinkler irrigation systems, where minimal leaching for salinity control is being practiced, leaching fractions of only 10 to 20% (based upon the amount of water applied) are not uncommon. With typical crop water needs throughout the irrigated West averaging 60 to 90 surface centimeters per year (with the exception of crops having unusually high demands for water, such as alfalfa), this means that only 6 to 20 surface centimeters of water need percolate through a well-managed, sprinkler-irrigated soil. For furrow-irrigated fields, on the other hand, variations in amounts of water applied from point to point on the landscape are considerably more pronounced. For some of the older irrigation projects of the Pacific Northwest, for example, it is not uncommon to have annual application rates (after conveyance losses have been deducted) of 120 to 180 cm of water. Hence, after crop water needs have been subtracted, one fourth to one half or more of the applied water is available for leaching of solutes (including nitrate) from the field. Fortunately, however, much of this water passes through previously-leached soil beneath the irrigation furrow, and is relatively ineffective in removing additional nitrogen from the crop root zone. In calculating the concentrations and total quantities (mass emissions) of nitrate from irrigated fields, it is necessary to have estimates of both the amounts of water being leached from a given field and of the corresponding concentrations of nitrate-nitrogen in that water. It is then a straightforward matter to multiply the two terms together in order to obtain an estimate of the quantity of nitrogen leaving the field.

In cases where a field has been tile-drained, it may be relatively easy to monitor the quantity of water leaving the field (although

Letey et al, 1977, found it virtually impossible in some settings), and to periodically monitor the nitrate concentration of the drainage waters (which normally changes only slowly with time), in order to obtain a measure of the mass emission of nitrate. One must be careful in such situations, however, that large quantities of water are not entering the drainage field from surrounding areas. This is particularly likely for undulating lands or where tiles drain relatively-level fields near the foot of surrounding slopes. It is also likely in cases where water is seeping from unlined canals in the vicinity. Such water commonly is quite low in nitrates, and can considerably dilute the concentrations of nitrogen actually leaving the crop root zone of the field in question.

In cases where a field is not tile-drained, the estimation of downward flux of water across the landscape becomes considerably more difficult. One cannot simply rely on the periodic monitoring of a few points in a field, for the points chosen may not adequately represent average water-flow characteristics of the particular soil, crop, and irrigation-system conditions which exist at the site. In addition, "pulses" of nitrate-nitrogen become less concentrated and more broadened as they move downward through the soil. Examples are provided in Figure 2. Such variations in pulse height and shape make intermittent sampling as hazardous with respect to time trends as with respect to positional trends across the landscape.

Crude estimates of the amount of water leaving the crop root zone can be obtained by measurement of the amount of water applied, coupled with estimation of the amount of water being used by the plant through micrometeorological measurements, evaporation pan estimates, or other types of crop consumptive use calculations. Estimates can also be obtained from lysimeter installations, where large, soil-filled cylinders are placed in the middle of a cropped field and either weighed continuously with an hydraulic device or evacuated periodically, with the amount of water leaving the cylinder then being trapped

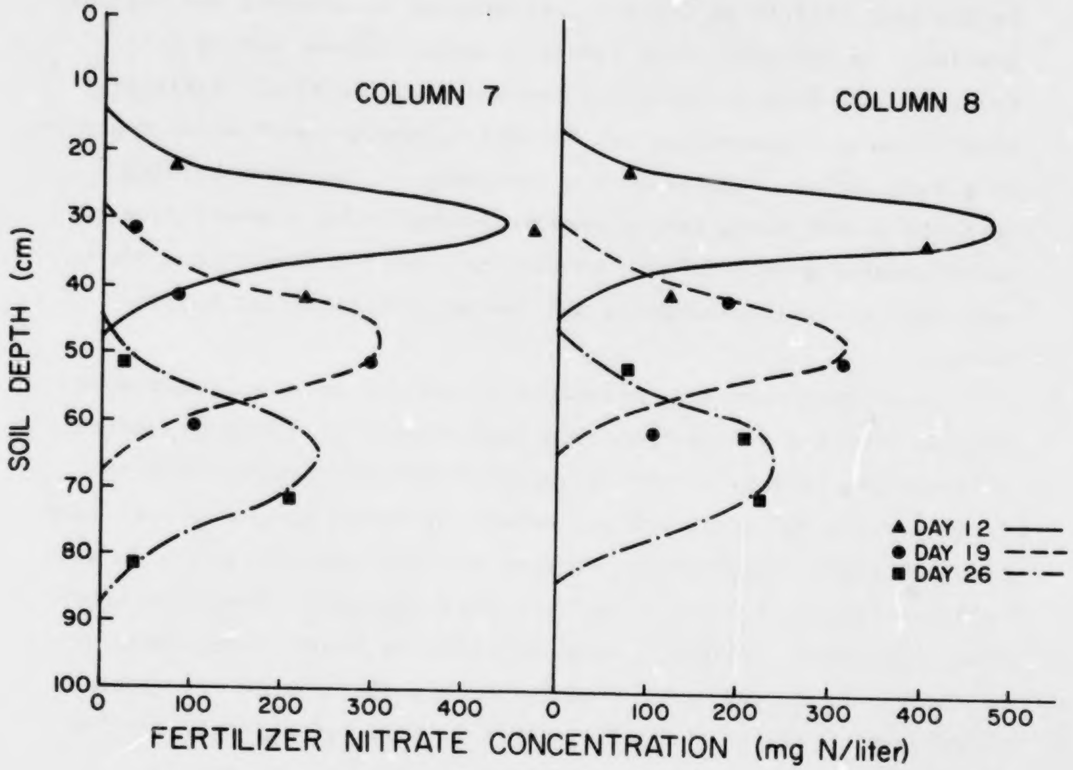

Figure 2. Concentration of fertilizer nitrate as a function of soil depth for three times after application of a nitrate pulse to two columns of Yolo loam topsoil. The solid and broken curves were calculated using appropriate denitrification rate constants (from Rolston and Marino, 1976).

and measured. However, it is difficult in many cases to exactly simulate crop growth and soil profile water-transmission properties with a lysimeter installation. For research purposes, estimates of downward water movement can be obtained from heavily-instrumented sites. Tensiometers, for example, can be placed at several depths in the soil profile in order to estimate water contents and hydraulic gradients in the soil, once curves of water content versus soil tensions have been obtained for the soils at the site. Similarly, neutron probe measurements can be used to monitor soil water contents. This information, coupled with a knowledge of the water-holding capacity of the soil, can be used to estimate the downward flux of water under prescribed field conditions. Such installations are extremely expensive, however, and are not practical for routine usage.

Several workers have attempted to monitor downward water movement in freely-draining fields through the use of nitrate/chloride ratios. The rationale for the approach is an assumption that the growing plant and soil microbes cause continued changes in soil solution nitrate concentrations, whereas chloride concentrations in many settings remain relatively unaffected by biological processes other than plant water uptake. Hence, if chloride is not being taken up by the plant, is not being produced by the dissolution of soluble chloride minerals in the soil, and is relatively-uniformly applied if any chloride-containing fertilizers are in use, changes in chloride concentrations should reflect only the removal of water by the transpiring plant. Pratt et al (1972) and Adriano et al (1972a, 1972b), for example, used relative chloride concentrations in the irrigation water and in the water of the unsaturated zone below the root zone to approximate the leaching fraction for fields they were monitoring. The approach worked particularly well for citrus, which almost quantitively excludes chloride. Changes in the nitrate/chloride ratio then reflect additional removal of nitrate from the system, as during crop nitrogen uptake or during losses of gaseous denitrification products from poorly-aerated soils. Corrections are required for crops like asparagus, which remove appreciable amounts of chloride from the soil-water system and adjustments must be made for the chloride added in fertilizers and in manures.

Early comparisons of relative nitrate and chloride movement, as by Wetselaar (1962), involved the monitoring of different salts in different soil profiles or at different points on the landscape. More recent work has centered around the use of ratios of different solutes for the same soil samples, however. For example, Kimble et al (1972) have also used chloride/nitrate ratios to delineate the amounts of denitrification occurring where manure has been applied as a nitrogen source. Once the amounts of water applied annually, and the leaching fractions, have been calculated, the downward water flux and the transit time through the unsaturated zone to the water table beneath an area can be estimated. Adriano et al (1972a) calculated transit times through the unsaturated zone beneath irrigated fields from the relation

$$\text{transit time in years} = \frac{S\theta}{D} \qquad (1)$$

where S = soil profile depth, θ = volumetric soil water content, and D = drainage water volume in surface cm/year. Using this relation, Pratt et al (1972) calculated transit times of 12 to 47 years for nitrate to move through a 30-meter unsaturated zone beneath southern California citrus groves, for an annual leaching fraction of approximately 0.4

In a similar vein, Pratt and Adriano (1973) related nitrate-nitrogen (NO_3^--N) concentrations in the unsaturated zone beneath irrigated fields to N_e (the excess nitrogen, defined as nitrogen inputs (N_i) minus nitrogen uptake (N_c) by the crop) and D (defined above) through the relation

$$NO_3^- \text{-N (ppm)} = 10\ N_e/D \qquad (2)$$

They reported the relation to be accurate to within ± 20% for fields where appreciable amounts of denitrification or of mineralization of organic nitrogen were not occurring. Unfortunately, the latter "exceptions" constituted 16 of the 25 settings reported in their paper.

In cases where the percolating waters are entering a relatively static and relatively deep groundwater body, the concentrations of nitrate-nitrogen in the percolating waters will be reflected by the concentrations of nitrate-nitrogen in the groundwater, or at least in the upper portions of the groundwater, as long as mixing remains incomplete. Where sprinkler irrigation is being employed, so that concentrations of nitrogen leaving the bottom of the root zone are at least hypothetically uniform across the field, the concentrations of nitrogen in the water leaving the bottom of the root zone will correspond roughly to the concentrations of nitrogen in the upper portions of the underlying groundwater body. In cases where a field is being furrow-irrigated, however, increments of water containing rather high concentrations of nitrogen will be diluted by increments of water that flow through the well leached portion of the soil under the furrow, and the groundwater body will in this case reflect the weighted average of the respective nitrate-nitrogen concentrations. Pratt et al (1972) have reported, in general, an inverse relation between the leaching fraction and nitrate-nitrogen concentrations of the unsaturated zone, although Pratt et al (1976) found this to be true only at higher rates of manure application for a set of field manure application plots, and Letey et al (1977) found no apparent relation between volume of tile drain outflow and drainage water nitrate concentration. In cases where the economics of nitrate loss are under consideration, where the leached nitrogen is being returned to surface streams or to rapidly-moving groundwater aquifers, or where numerous sources are contributing to the nitrogen levels of a groundwater body, the total quantity of nitrogen being emitted is of considerably greater importance than the actual concentrations of nitrate involved. This point has been stressed strongly in the conclusions of Letey et al (1977), who point out that the situation is not unlike that of leaching for salinity control, where the smallest total quantities of salt are returned to receiving bodies whenever the salt is concentrated into as small a volume of water as possible. The same trends have been reported for nitrates (Devitt et al, 1976; Pratt et al, 1976). Whether one is concerned with the concentrations or with the total quantity (mass emissions) of nitrate leaving a given irrigated tract largely depends on the ultimate fate of the nitrogen, on the purposes for which the monitoring program is being conducted, and on the extent to which single vs. multiple sources of nitrogen for the receiving body need be considered.

IRRIGATED CROPLAND

According to Stanford (1973), corn grown under good management commonly recovers 50 to 70% of the nitrogen applied, with the rest being largely immobilized during residue decomposition, and hence unavailable for nitrate leaching. He acknowledged that much nitrogen leaches or denitrifies at higher application rates, however. Data such as those provided by Nelson and MacGregor (1973) and MacGregor et al (1974) confirm such predictions, with little residual soil nitrate accumulating in long-term corn plots until nitrogen application rates reached nearly 200 kg nitrogen/ha/yr, at which rate residual soil nitrate levels increased on the order of three- to five-fold. Similar results were reported by Schuman et al (1975) as long as nitrogen fertilization rates for corn remained less than 170 kg nitrogen/ha/yr, though Ludwick et al (1976) reported residual soil nitrate-nitrogen levels to increase linearly with nitrogen fertilization rate at rates ranging from 70 to 270 kg nitrogen/ha/yr. In the latter studies, it was concluded from data for 270 irrigated farms in central and eastern Colorado that residual nitrate can accumulate in irrigated profiles even if nitrogen has been applied at rates less than required for optimal crop production.

Pratt et al (1972) monitored long-term research plots and commercial citrus groves in southern California, and concluded that nitrate-nitrogen concentrations in the unsaturated zone beneath well-managed groves were being kept below acceptable levels if the fields were managed for maximal crop production and if a reasonably high leaching fraction was being maintained for salinity control. If nitrogen application rates were not excessive, and if the soils were porous and had no water-impeding textural discontinuities, nitrate-nitrogen concentrations in the unsaturated zone could be predicted adequately from equation 2. Where impeding layers occurred in the top 60 to 90 cm of soil at two sites, however, nitrogen losses (most probably due to denitrification) averaged 50% of nitrogen inputs.

For a nearby watershed from which about half of the applied water emerged as drainage water, Bingham et al (1971) found nearly half of the fertilizer nitrogen to be leached each year.

Results of nitrate leaching studies reported by Adriano et al (1972a), Adriano et al (1972b), and Pratt and Adriano (1973) are summarized in Table 1. As annual nitrogen inputs were increased, the most consistent change was in terms of the decreased percentage of nitrogen reaching the crop, along with consistent (though not proportional) increases in nitrate-nitrogen levels of the unsaturated zone beneath the fields. An exception to the latter trend was for citrus, where growers seemed to be managing their nitrogen more consistently than for any of the other crop groups. Leaching losses of nitrogen commonly averaged 25 to 50% of the nitrogen applied in most cropping situation. In cases where large percentages of nitrogen were being leached, grower inputs such as improved water and fertilizer management, as well as possible changes in water application method, could reduce losses of fertilizer nitrogen (and hence nitrogen requirements) substantially.

Table 1. Data for nitrogen inputs, nitrogen balance, and measured nitrate-nitrogen concentrations in waters of the unsaturated zone beneath some typical southern California croplands (recalculated from Pratt and Adriano, 1973).

Crop	No. Sites	Nitrogen Inputs (kg/ha/yr)	Nitrogen in Crop	Nitrogen Leached	Nitrogen Losses	Nitrate-nitrogen Concentration (mg/l)
			%	%	%	
Citrus	7	111-194	21-34	22-91	-17 to 56	21-70
Citrus	2	256-330	11-16	43-48	36-47	29-64
Citrus	1	414	10	36	54	47
Asparagus	2	130-144	62-69	19-29	9-12	12-14
Asparagus	2	478-492	27-31	23-28	45-47	29-90
Celery	1	385	32	58	9	35
Celery	1	1663	14	29	58	78
Misc. Row Crops	2	210-215	70-71	40-102	-10 to -73	56-60
Misc. Crops	5	360-530	15-44	13-71	1-55	36-123
Misc. Crops	1	740	31	76	-6	85
Misc. Crops	1	1525	25	60	15	120

In cases where substantial nitrogen losses (e.g., through denitrification) were recorded, factors such as water-impeding layers within the crop root zone may well have been beyond grower control. Cases where substantial negative "losses" were recorded generally corresponded to sites where nitrogen was being mineralized from former manure applications or from residual soil nitrogen on the site. Adriano et al (1972a) concluded that growers in the Santa Ana River Basin were using about 135 kg nitrogen/ha/yr above that removed by the harvested crop at the time of these studies. In cases where no denitrification losses occurred (e.g., where no restrictive layer impeded water movement in the surface 2 meters of depth), this would lead to nitrate-nitrogen levels in the unsaturated zone of 20 to 40 ppm for the 30 to 60 cm leaching volumes common to this area. Adriano et al (1972b) concluded that crops consistently remove less than 50% of the nitrogen applied throughout the southern California area. For asparagus, they reported 350 to 500 kg nitrogen/ha/yr being applied routinely to crops needing only 200 to 250 kg nitrogen/ha/yr, and that levels as high as 1000 to 2000 kg/ha have accumulated in 15 meter profiles below the most heavily-fertilized fields. A 36 cm leaching volume was reported as common for this crop. For celery, it was reported as common to apply 450 to 500 kg nitrogen/ha/yr to crops requiring only 300 to 350 kg nitrogen/ha/yr. Highest denitrification and immobilization losses in this case (up to 68% of the nitrogen applied) were from plots to which part of the nitrogen had been supplied in the form of chicken manure. On the other end of the spectrum, Letey et al (1977) reported extremely low losses of nitrate-nitrogen from alfalfa fields receiving virtually no nitrogen fertilizer applications (with crop nitrogen needs being supplied by nitrogen fixed from the atmosphere by this leguminous crop).

Relation of unsaturated-zone nitrate-nitrogen levels to the texture of the hydraulic control section of the overlying soil profile has been demonstrated by Lund et al (1974) and Devitt et al (1976). Lund et al showed that 86% of the variability in nitrate-nitrogen levels of the lower profile could be related to texture of the hydraulic control section, with lower nitrate-nitrogen concentrations evident wherever there was some accompanying restriction to water movement through the profile. Their results are illustrated in Figure 3, where lower-profile nitrate-nitrogen levels are seen to be well correlated with clay content of the hydraulic control section. In a similar vein, Devitt et al (1976) found very low denitrification potentials in coarse-textured profiles, with chloride/nitrate ratios indicating appreciable denitrification only in cases where high-clay subsurface layers occurred. Their only reported exception to a high-clay, low nitrate-nitrogen trend occurred for one profile which had high clay content, and also high initial residual nitrogen levels, throughout. Similar relation of denitrification losses to soil profile clay contents were reported by Letey et al (1977).

Many workers regard nitrate-nitrogen which has leached beyond the top 2 meters of soil as beyond the zone of significant denitrification, although Meek et al (1970) suggested substantial reductions in nitrate-nitrogen levels of drain-tile effluents if the tile lines could be submerged in order to promote denitrification.

Nitrogen levels in the root zone during and following average growing seasons, as a function of both water and fertilizer management practices, have been provided from studies from the Columbia Basin area of central Washington (McNeal and Carlile, 1976). Typical results are summarized in Table 2, for potato fields of that area. As is evident from the data, concentrations of nitrate-nitrogen in the lower root zone and subsoil were commonly 3 to 10 times the drinking water standard. Hence, the displacement of any water from the root zone would lead to substantial losses of nitrogen. In a case where the actual amount of water leaving the root zone was monitored for a growing season from tensiometer and water content data, the drainage volume amounted to 12 surface cm for a 123-day growing season, or 59 kg nitrogen/ha at an average dissolved-nitrogen concentration of 50 ppm. This was for a well-managed, sprinkler-irrigated plot, by the way, and considerably greater amounts of nitrogen would normally be leached under furrow irrigation. However, the difference would not be as great as predicted from rela-

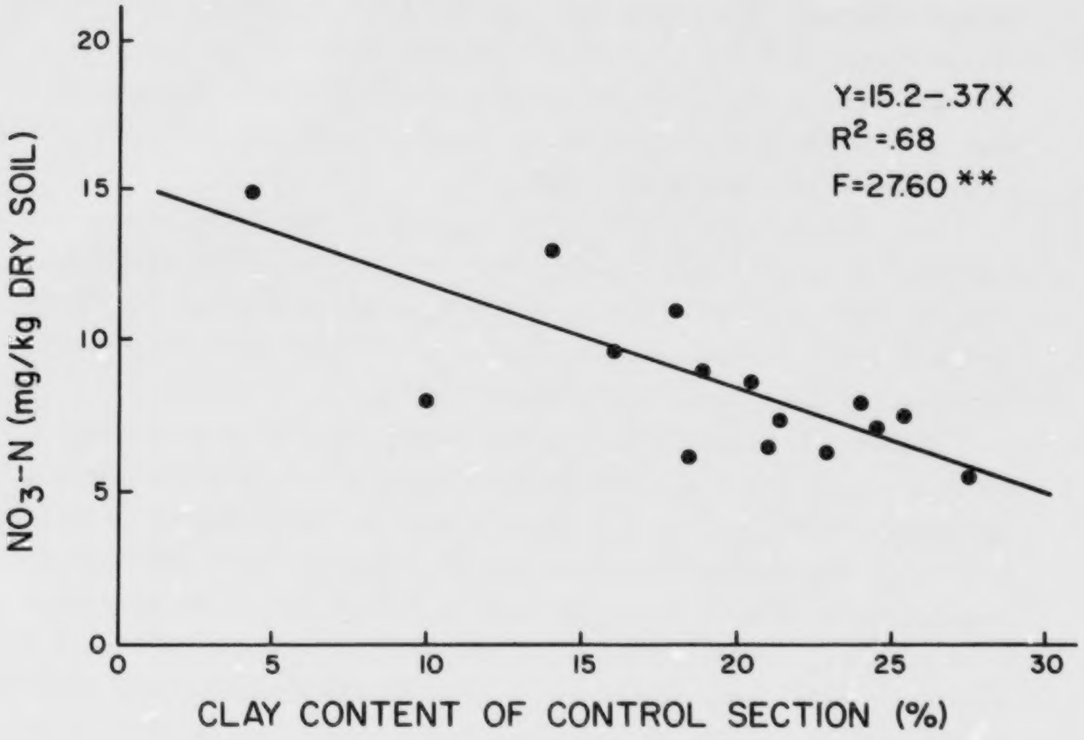

Figure 3. Relationship between average subsoil nitrate concentrations (1.8 to 8 m soil depths) and average clay content of the control section of some soil profiles (from Lund et al, 1974).

Table 2. Nitrate-nitrogen concentrations within and beneath potato root zones in the Columbia Basin for various irrigation systems.

Experiment	Soil Type	Depth (cm)	Nitrate-nitrogen Concentration (mg/l)			
End-of-Season Values						
			Sprinkler Rate		Furrow Rate	
			Low	High	Low	High
Irrigation rate study	Sand	0-120	298	54	26	2
		120+	78	33	12	3
			Head of Field		Tail of Field	
			Low Rate	High Rate	Low Rate	High Rate
Furrow irrigation	Silt loam	0-60	38	30	104	44
		60+	13	6	93	31
Within-Season Values						
			Early Season		Late Season	
			Low Rate	High Rate	Low Rate	High Rate
Sprinkler irrigation	Silt loam	0-60	436	612	88	24
		60+	68	71	102	61
			110 kg N/ha	450 kg N/ha	100 kg N/ha	450 kg N/ha
Grower's field center-pivot	Silt loam	0-60	193	461	121	178
		60+	52	48	39	68
			250 kg N/ha	520 kg N/ha	250 kg N/ha	520 kg N/ha
Grower's field center-pivot	Sand	0-60	177	204	48	154
		60+	56	68	61	127
			110 kg N/ha	450 kg N/ha	110 kg N/ha	450 kg N/ha
Grower's field alternate-furrow	Sandy loam	0-60	107	248	53	130
		60+	10	22	7	11
Grower's field solid-set sprinkler	Silt loam	0-60	155	367	43	101
		60+	95	121	69	111

tive drainage volumes of sprinkler-irrigated and furrow-irrigated fields. There would also be considerably more low-nitrate water being leached under furrow irrigation, as water passed through previously-leached soil immediately beneath the irrigation furrow.

One of the factors accounting for high nitrate leaching potentials for potato fields is the fact that the potato is a crop having atypically-high water requirement. If the amount of water available to potato roots decreases appreciably, older leaves will senesce and the amount of photosynthetic surface will be decreased. Hence, high soil water contents must be maintained if one is to maximize crop production potential. In addition, potatoes commonly are grown on rather sandy soils, so the combination of high water application rates and low soil water-holding capacities leads to considerable potential for leaching. In addition, the potato does not seem to be able to utilize soil-solution nitrogen nearly as efficiently as does a crop such as wheat. In the wheat-growing region of eastern Washington, for example, it is common to find virtually no nitrate-nitrogen remaining in the surface 180 cm of soil by the end of the growing season, if sufficient water has been present to realize maximal crop growth potential. Under controlled growth conditions, Warncke and Barber (1974) reported that sorghum, soybeans and brome-grass reduced root-zone nitrate-nitrogen levels to the range 0.02- to 0.04 mg/l. In a well-managed potato field, on the other hand, it is common to find 20 to 40 mg/l of nitrate-nitrogen remaining in the soil solution at the end of the growing season (McNeal and Carlile, 1976). Kirkham et al (1974) reported a situation where 50 mg/l nitrate nitrogen was insufficient for continued potato growth on a rather sandy soil. Hence, crop utilization efficiency must be considered when evaluating pollution potential or ability of selected management practices to reduce nitrate-nitrogen losses from croplands. In cases where crops are to be grown which are inefficient nitrogen utilizers, it may be necessary to put almost complete stress on proper water management, and on subsequent growth of deep-rooted crops to "scavenge" residual nitrate-nitrogen from the soil profile (Mathers et al, 1975), if pollution from nitrate leaching is to be minimized.

As is evident from Table 3, the concentrations of dissolved nitrogen in the root zone vary markedly from early season to late season. Hence, the passage of excess water through the soil profile in early season can be particularly hazardous, as in cases where the water application rate is improperly adjusted to plant growth needs or where additional water is applied for wind-erosion control on sandy soils. It should also be noted from the table that concentrations of nitrate-nitrogen beneath the crop root zone in a furrow-irrigated, sandy loam soil were quite low throughout the growing season. This is probably because nitrate was continually being leached by the excessive amounts of water passing through the soil beneath the furrows under these conditions. The alternate-furrow irrigation approach, as used at this location and as commonly employed throughout the Columbia Basin, tends to "trap" nitrogen in the crop root zone, for the nitrogen is moved back and forth between furrows, rather than being leached from soil beneath the furrows, during irrigation. A comparison of the alternate-furrow data of Table 3 for the sandy loam site to that for the silt loam site shows that soil texture has a marked effect on the amounts of nitrogen persisting in the crop root zone and in the underlying subsoil. In the latter case, because of smaller amounts of water moving downward during irrigation, the amounts of nitrogen remaining even in the soil beneath the crop root zone were substantial throughout the growing season.

A common problem associated with the growth of potatoes in the Columbia Basin is the fact that there appear to be excessive amounts of nitrogen recommended for growth of the potato crop. In some cases, nitrogen recommendations were developed originally for lands which had been only recently brought into production. Such recommendations have been maintained, even though potato yields commonly decrease on

Table 3. Nitrate-nitrogen concentrations within and beneath potato root zones in the Columbia Basin for various fertilizer treatments (all values in mg/l).

Experiment	Soil Type	Depth (cm)	Nitrate-nitrogen Concentration (mg/l)			
End-of-Season Values			Broadcast		Banded	
			110 kg N/ha	670 kg N/ha	110 kg N/ha	670 kg N/ha
Suspension Fertilizers (1972)	Silt loam	0-60	37	89	38	54
		60+	21	43	23	42
Following-Season Values			Spring of 1972		Fall of 1972	
			110 kg N/ha	670 kg N/ha	110 kg N/ha	670 kg N/ha
Suspension Fertilizers (1971 Expt. Sampled in 1972)	Silt loam	0-60	80	174	15	27
		60+	32	105	39	67
Within-Season Values			Early Season		Late Season	
			Traditional	Slow N	Traditional	Slow N
Grower's center-pivot (Slow-release N)	Sand	0-60	469	352	150	226
		60+	83	51	105	64
Grower's alternate-furrow (Slow-release N)	Sandy loam	0-60	256	128	95	94
		60+	18	9	14	12
Grower's solid-set sprinkler (Slow-release N)	Silt loam	0-60	296	162	85	146
		60+	84	50	71	57
Alternate-furrow (Slow-release N)	Silt loam	0-60	354	265	236	154
		60+	81	23	100	32
			340 kg N/ha	670 kg N/ha	340 kg N/ha	670 kg N/ha
"Nitrogation" (Sprinkler-applied N)	Silt loam	0-60	102	33	76	89
		60+	71	17	40	22

recropped potato lands because of plant disease problems. Because of the decreased crop yields on recropped lands, and also because of the substantial amounts of nitrogen released from potato crop residues during the following season on such lands, the amounts of nitrogen required are considerably less under recropped conditions.

Information has also been obtained from the Columbia Basin studies suggesting that the concentrations of nitrogen recommended to be in plant tissues throughout the growing season may in some cases be excessive. It might be desirable to retest some of the plant tissue needs against crop yields periodically, or whenever fertilizer management changes markedly. With respect to nitrogen, for example, the traditional approach to nitrogen management was to put on virtually all of the required nitrogen at the start of the growing season, so that high tissue nitrogen levels were common in early season, with low levels being attained by mid- to late-season, as the nitrogen supply became exhausted. Recommended tissue nitrogen levels were largely developed from this particular approach to nitrogen fertilizer management. In recent years, however, more and more of the nitrogen has been applied in the irrigation waters during the growing season, so it may no longer be necessary to have the high early-season nitrogen levels which were once thought necessary. Attempts to maintain such levels may be contributing somewhat to an over-supply of nitrogen to the potato crop in this area, and hence to a nitrate leaching problem.

Unfortunately, the determination of actual crop nitrogen needs under constantly-changing growing conditions is difficult for most growers and even for many farm management services. These people, faced with the need to make management decisions throughout the growing season, prefer to keep the crop both heavily watered and heavily fertilized, in an attempt to maximize economic production. In relatively few cases are adequate field plot data available as new management techniques become popular for a given area. These field plots are essential if one is to demonstrate whether the amounts of nitrogen that were required formerly will still be needed under the new set of conditions or if new criteria are needed.

Some management alternatives for preventing deep percolation losses of excess nitrogen are listed in Table 4. Possible constraints to the adoption of each alternative are also listed in the table.

Table 4. Some management alternatives for preventing leaching losses of excess nitrogen.

Management Alternatives	Possible Constraints
1. Use recommended nitrogen fertilization rates.	May be leaching more nitrogen than normal, or having substantial denitrification. Soil solution nitrate levels may not be reduced in proportion to decreased fertilizer usage.
2. Avoid heavy early-season irrigations, unless ammonium nitrogen is present or split nitrogen applications are used.	Increased wind erosion, more expense if several light irrigations are substituted.
3. Avoid single large nitrogen applications, while using more split nitrogen applications, or applying more nitrogen via the irrigation water.	May be more expensive for multiple nitrogen applications, or for metered nitrogen sources.
4. Use slow-release nitrogen sources	Increased fertilizer costs.

Table 4. Continued

Management Alternatives	Possible Constraints
or nitrification inhibitors.	
5. Properly sample and allow for residual, crop residue, and mineralizable soil nitrogen.	Increased sampling and analysis costs, uncertainty of nitrogen release rates.
6. Convert from furrow- to sprinkler-irrigation systems.	Increased capital outlay, increased energy demands.
7. Use light, frequent irrigations to keep nitrogen in the crop root zone.	Increased labor costs, loss of grower freedom for other operations, less reserve in case of marked changes in crop water demand.
8. Irrigate at a rate approximating crop evapotranspirative needs.	More rigorous demands on the irrigator, insufficient water where sprinkler patterns are wind-distorted or near the tail of furrow-irrigated fields.
9. Strive for increased water-application uniformity, and adequate levels of other nutrients, in order to maximize crop yields and crop nitrogen uptake.	Increased irrigation-system costs, or increased demands on the irrigator.
10. Minimize the amount of excess	Increased costs of improved

Table 4. Continued

	Management Alternatives	Possible Constraints
	water applied, consistent with requirements for salinity control.	water management, less uniform crop appearance and yields because of non-uniformities in water application.
11.	Promote development of deeper crop root zones, or periodically plant more deeply-rooted crops in the rotation.	Sufficiently deep-rooted crops not always available, and decreased flexibility or decreased economic returns may result from the crop rotation.
12.	Avoid excessive nitrogen leaching at the end of the cropping season, or plant winter cover crops which will use water and nitrogen.	Increased costs of additional monitoring or irrigation management, increased difficulty of planting in cover-crop residues.
13.	Permit less thorough aquatic weed control in return-flow ditches.	Increased water usage for the system.
14.	Use more alternate-furrow irrigation in gravity-irrigated areas.	Slightly increased irrigation management costs.
15.	Refine crop nitrogen recommendations as changes occur in crop management practices or average crop yields for the area.	Necessary correlative work is both time-consuming and costly.
16.	More carefully monitor and regulate amounts of water applied.	Both time consuming and labor-intensive.

Table 4. Continued

	Management Alternatives	Possible Constraints
17.	Try to grow crops that are relatively efficient utilizers of nitrogen, whenever possible.	Limited crop selection, possible economic losses when using more nitrogen-efficient crops.

If high nitrate concentrations arise or persist:

1. Limit domestic use of such waters.
2. Provide alternate water supplies for susceptible persons.
3. Adequate publicity and/or warnings in more seriously-affected areas.
4. Dilute or mix high-nitrate waters with low-nitrate waters.
5. Use adequate sampling programs when trying to isolate particularly-severe nitrate sources.

IRRIGATED WASTE DISPOSAL SITES

Animal and municipal waste disposal sites often offer almost singular potential for nitrate leaching, because of the high nitrogen loading rates which are commonly employed. The amounts of nitrogen contained in wastes, though variable with storage and pretreatment, is generally appreciable, so that annual waste loadings at a particular site are often based upon the amount of nitrogen that will be utilized by the crop growing on the site, plus the amount of nitrogen that can be safely estimated to be denitrified at the site during the waste disposal season. Fortunately, the amounts of denitrification at waste disposal sites are generally substantial, because of the soluble carbon loadings and the rapid oxygen depletion that is common as the active microbial population at the site oxidizes organic components from the wastes. A combination of high soluble carbon and nitrate loadings, high microbial activity, and high water contents or rapid oxygen depletion due to microbial activity present near-optimal conditions for the denitrification process.

For example, Kimble et al (1972) found more nitrate leached from ammonium nitrate fertilizer than from dairy manure of approximately equivalent nitrogen content, and King and Morris (1972) found that more than 50% of the nitrogen applied as sewage sludge stayed in a surface crust of sludge, and much of the remainder was denitrified, so that little nitrate pollution arose even from sludge application rates of 1000-2000 kg nitrogen/ha/yr. At the same site, nitrate pollution from commercial fertilizers began at 360-490 kg nitrogen/ha/yr. Similarly, Smith (1976) determined that only about 10 kg nitrogen/ha/yr entered a water table at the 150-cm soil depth beneath plots to which over 1000 kg nitrogen/ha/yr had been applied as food processing wastes, in a setting where 360 kg nitrogen/ha/yr was removed by a grass crop. In each of the last two cases, denitrification losses were 2 to 4 times crop removal losses under well-managed waste disposal conditions.

Meek et al (1974) must have achieved some type of a record with the application of 180-360 tons of manure/ha to Holtville clay under irrigated conditions without appreciable increase in subsoil nitrate-nitrogen concentrations at the 140-cm soil depth, although soluble carbon moved to at least 80 cm in one set of plots. A more realistic manure loading rate was probably that of Mathers and Stewart (1974), who found that feedlot manure could be applied to corn at rates of up to 22 tons/ha (equivalent to 300 kg nitrogen/ha/yr) without appreciable nitrate-nitrogen leaching, although manure loading rates of 112 to 240 tons/ha/yr gave nitrate-nitrogen levels of over 1000 kg/ha to the 180-cm soil depth. Pratt et al (1976) found appreciable nitrate-nitrogen leaching at 21 tons/ha/yr for a liquid manure averaging 4.5% nitrogen, and at 40 tons/ha/yr for a dry manure averaging 1.6% nitrogen. Mathers et al (1975) also demonstrated the effectiveness of alfalfa in removing water and nitrate-nitrogen from manure-contaminated plots to a depth of 3.6 meters in 2 seasons, at a seasonal nitrate-nitrogen removal rate of 300 kg nitrogen/ha. Adriano et al (1971) demonstrated nitrate-nitrogen flux to the groundwater which was approximately 3 to 3.5 times higher for corral sites than for pasture or cropland, but they also provided perspective with the statement that the ratio of cropland to corral areas in their study locale was nearly 13 to 1.

From the above evidence we can conclude that it is not unusual for denitrification from waste disposal sites to reach levels of 50 to 75% of the applied nitrogen. It should be kept in mind that the denitrification potential of a site may not be fully realized, however, if the site is maintained relatively dry (hence decreasing both microbial activity and the tendency toward anaerobic conditions), is relatively cold (because of decreased microbial activity), or is used for wastes which contain sources of soluble carbon that are only slowly utilized as an energy source by soil micro-organisms. Mineralization of nitrogen from wastes applied in former years must also be considered in deciding on proper waste loading rates for a given site. For example, Pratt et al (1976) found 45% and 75% of the nitrogen in a dry dairy manure and a liquid feedlot manure, respectively, to be available the first year after application, with decay functions approximating 10-15% release in the second year after application and 5% in the third and fourth years. In some respects the manager of a waste disposal site may be regarded as a "conductor", as he attempts to orchestrate conditions at the site which will lead to as much denitrification as possible, so that the amounts of nitrate leaching from the site will be minimized.

PREDICTION OF NITRATE LEACHING LOSSES

Nitrate leaching loss values from several of the southern California studies have been described by the empirical equation

$$N_L = 0.20\ (N_i D)^{0.712} \qquad (3)$$

where N_L is nitrogen leached (kg/ha/yr), N_i is nitrogen inputs (kg/ha/yr), and D is drainage water volume (surface cm/ha/yr). Although using a different descriptive equation, Letey et al (1977) also found the total quantity of nitrogen leached from tile-drained fields to be well correlated with total water discharged and with fertilizer nitrogen application rates. As is evident from Figure 4, the relationship adequately described a wide range of data for cropped fields and for manure field plots in the Santa Ana River Basin and in the coastal and interior valleys of California. The correlation coefficient relating predicted and measured leaching losses was 0.95, with a correlation coefficient of 1.0 indicating perfect correlation between sets of values. Because of the marked effect of the largest numbers on a regression such as this, however, the correlation coefficient would have been considerably lower had the data been restricted to the 0-200 or 0-300 kg nitrogen/ha range.

Pratt et al (1977) have subsequently shown that predictions of nitrate leaching can be improved significantly by incorporating the saturated hydraulic conductivity of undisturbed cores of surface soil into the predictive equation. Such incorporation accounts for variations in denitrification as drainage becomes impeded at higher soil clay contents. Their best predictive equation had the form

$$N_L = 22.3 + 0.195\ NiH + 0.0337D^2 \tag{4}$$

where H is saturated soil hydraulic conductivity (cm/hr) and all other terms are as described previously. Because of the strong positive influence of soil hydraulic conductivity on nitrate leaching from croplands, Pratt has even suggested that surface soil hydraulic conductivity might be lowered in some cases to promote denitrification and decrease nitrate leaching losses.

When the approach of equation 3 was applied to data from furrow-irrigated portions of the Pacific Northwest, the equation appeared to significantly over-estimate the amounts of nitrogen being leached.

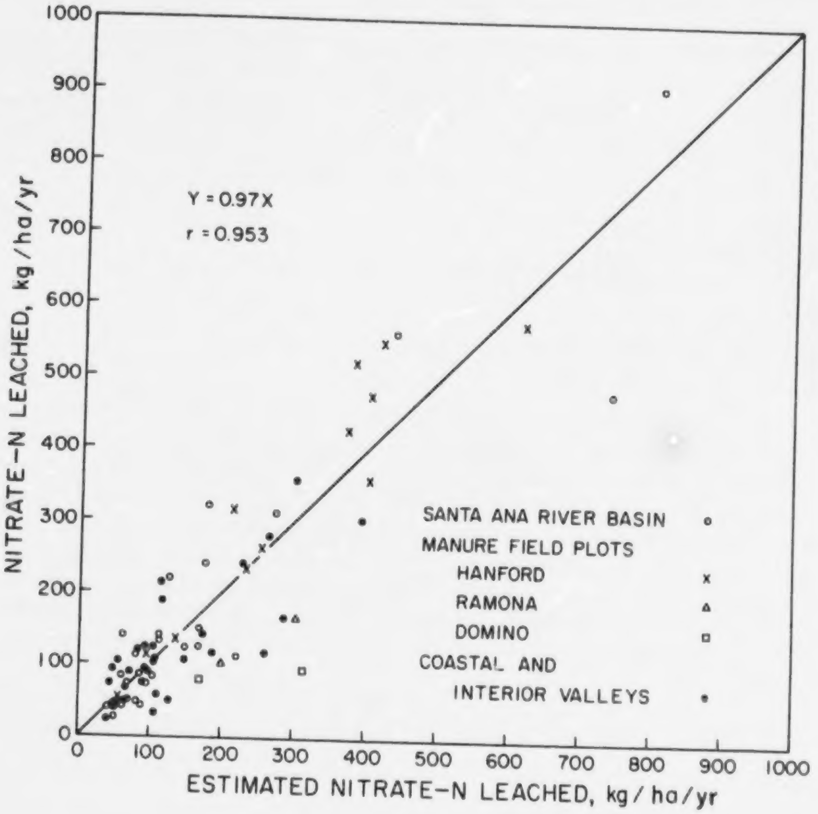

Figure 4. Relationship between measured and estimated nitrate leaching. The estimated values are from the equation $N_L = 0.20(NiD)^{0.712}$.

This comparison is provided in Table 5.

Table 5. Field-testing of nitrate leaching predictions for two Pacific Northwest irrigated tracts.

Irrigation Tract	Workers	Measured Losses	Predicted Losses*	Predicted/ Measured
		--kg nitrogen/ha/yr---		
Wapato, Washington	Nelson and Weaver (1971)	31.8	95.2	3.0
Twin Falls, Idaho	Carter et al (1971)	33.0	70.6	2.1

* Based on equation 3, using crop acreage and average leaching volume figures provided by the authors, and crop fertilization rates conforming to current fertilizer guides for the respective areas. Carter's figure of subsurface return-flows averaging 50% of water diverted, and Twin Falls tract estimates of 30% cumulative project-wide and on-farm conveyance losses (D. L. Carter, personal communication), which would constitute part of the subsurface return flows, used for both tracts.

As is evident from the table, the amount of nitrogen predicted to be lost from these irrigated areas through use of equation 3 exceeded actual losses by a factor of 2- to 3-fold. Although part of the discrepancy could be due to enhanced denitrification under Pacific Northwest conditions, a more probable reason for this discrepancy is the fact that furrow-irrigated fields have considerably larger amounts of water passing rather harmlessly through the root zone once the soil beneath the irrigated furrow has been leached. In other words, incorporation of the entire volume of drainage water weights the values too heavily in relation to the amounts of nitrogen which would actually be leached per unit of water transmitted through the soil.

Attempts are being made at present to improve the leaching prediction by incorporating coefficients of uniformity for various types of irrigation systems. Hence, furrow-irrigated systems, with their lower coefficients of uniformity (in other words with a greater amount of heterogenity of water application), should be expected to have lower leaching estimates from the equation than sprinkler-irrigated fields, with considerably higher uniformity coefficients. It should be kept in mind in this regard that extreme inhomogenities in water movement have been reported even for sprinkler-irrigated California fields, and have been verified under Columbia Basin conditions (McNeal and Carlile, 1976). Hence the variations for furrow-irrigated fields are truly enormous.

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EFFECT OF WATER MANAGEMENT ON NITRATE LEACHING

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ABSTRACT

Nitrates which are leached (transported below the root zone) represent a resource loss and a potential contribution to water pollution. The amounts of leached nitrates for a given time period were determined at various commercial farming sites in California and in a carefully controlled experimental plot receiving various water and fertilizer application treatments. Some of the agricultural sites had tile drainage systems and others had "free drainage" to the groundwater. Linear regression analyses were conducted on the data. Similar results were observed for the tile and free drainage systems. The highest correlation coefficient was achieved for the amount of leached nitrates versus the product of the drainage volume and fertilizer nitrogen application. The next highest correlation coefficient was for amount leached versus drainage volume followed by amount leached versus fertilizer nitrogen application. In most cases there was no significant correlation between nitrate concentration in the water below the root zone and drainage volume or fertilizer nitrogen application. A significant linear relationship between amount of leached nitrate and drainage volume was also obtained at the experimental plot.

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INTRODUCTION

There are two concerns with nitrate leaching (the term "nitrate leaching" will be used to represent the process of transportation of nitrates below the root zone). First, nitrate leaching represents a resource loss. Nitrogen is an essential element for plant growth, and crop production is usually increased by nitrogen addition to the root environment. Leaching of nitrogen beyond the root zone, therefore, represents the waste of an important resource. Secondly, nitrate can be considered a water pollutant under certain circumstances. Nitrogen in water can contribute to eutrophication and stimulate vigorous biological activity when the activity is undesirable. High nitrate concentration in water is also considered to be a health hazard when given to infants. Waters that leach nitrates may become surface waters through natural or artificial drainage systems or move to the ground water which will be later pumped to the surface for various uses.

Minimizing nitrate leaching is a worthy goal because it conserves a valuable resource and minimizes the hazard of water pollution. The purpose of this report is to evaluate the relationships between water management in irrigated agricultural systems and nitrate leaching.

The term "water management" encompasses many facets. It can include consideration of timing and amount of water application. It also includes consideration of application methods such as furrow, sprinkler, border or drip irrigation. Each of these water management features conceivably affects nitrate leaching. The paper by McNeal and Pratt in this volume considered some aspects of this subject. This report will be restricted to presenting data obtained by the University of California on a project supported by the RANN Division of the National Science Foundation and the University of California Kearney Foundation of Soil Science. We do not presently have data which would allow us to evaluate the effects of irrigation application techniques such as furrow or sprinkler systems on nitrate leaching. Indeed, the present data is restricted to considering the amount of water application or, more specifically, the amount of water leached beyond the root zone.

It is necessary to have a quantitative expression of nitrate leaching to report research findings. Two parameters may be used. The nitrate-nitrogen concentration in the water at a depth below the root zone may be used as one index of nitrate-nitrogen leaching. A second parameter is the quantity of nitrate-nitrogen which has moved beyond the root zone in a given time period. The latter parameter is typically referred to as mass emission. The question immediately arises as to which is the preferable parameter to properly characterize nitrate leaching.

Mass emission is clearly the more significant parameter when resource loss is of prime consideration. An increase in mass emission represents an increased loss of nitrogen resource as a fertilizer element. The appropriate parameter is not so obvious in considering environmental pollution. Biological systems usually respond to concentrations of nitrate-nitrogen. For example, the present U.S. Public Health standard is 10 ppm nitrate-nitrogen in drinking waters. Thus, nitrate-nitrogen concentration is of importance from an environmental point of view. On the other hand, when a potential pollutant is discharged into the environment, its negative effect depends on the assimilative capacity of the environment for that constituent. The environment is negatively impacted when the amount discharged exceeds the assimilative capacity. In this respect, mass emission is a better parameter than concentration. Clearly increasing the mass emissions increases the probability that the assimilative capacity of the environment will be exceeded.

A very important point to recognize is that mass emission and concentration are not generally proportional in agricultural systems.

Indeed, it is not unusual for mass emission and concentration to be inversely related. This is in contrast to the case for water discharge from municipalities or industries. For municipalities or industries, concentration is directly proportional to mass emission because the volume of water to be discharged is relatively constant. Decreasing the concentration of a constituent in a fixed volume of water by 50% results in a 50% decrease in mass emission. Thus, regardless of whether concentration or mass emission is the better criteria for controlling pollution of the environment, the same conclusions are drawn for the industries or municipalities. We will report most of our nitrate leaching both in terms of mass emission and concentration.

EXPERIMENTAL APPROACH

Three research activites are reviewed. One is referred to as the "free drainage" investigations which were carried out under the direction of Drs. P. F. Pratt and J. Rible. Another research activity is referred to as "tile drainage", studies carried out under the direction of Dr. Letey. The third activity is referred to as the "Davis plots". Only a small part of the research carried out on experimental plots located near the Davis campus will be given in this report. The data presented in this report were provided by Drs. J. W. Biggar, D. R. Nielsen, and S. Simmons.

The "free drainage" studies involved drilling on specifically chosen agricultural sites and analyzing the soil removed at various depths. The depth of drilling was 50 or more feet except in cases where mechanical difficulties with sand or rocks prevented further digging or a saturated zone was reached.

The concentration values reported are averages from all samples collected from the 15-foot depth to the maximum depth of sampling at a given site. In order to calculate mass emission a value for the amount of water moving beyond the root zone in a given time is also required. The following procedure was used to estimate the drainage volume. The leaching fraction was estimated from the ratio of the chloride concentration in the irrigation water to the average chloride concentration in the soil water below the root zone. This procedure assumes that the only chloride input was that in the irrigation water and that no significant amount of chloride was removed by the crop. Thus the increase in chloride concentration observed in the soil water below the root zone was a result of water lost from evapotranspiration. Multiplication of the leaching fraction by the amount of water applied produced a value for the drainage volume. The calculated drainage volume was then multiplied by the nitrate-nitrogen concentration in the water beyond the root zone to estimate mass emission.

Tile drainage effluents were collected from commercial farming operations at various areas in California where tile drainage systems are common. Tile effluent samples were usually collected weekly and analyzed for nitrate-nitrogen concentration. The tile flow rate was measured when possible by collecting water in a bucket for a given period of time and measuring the collected volume. Tile flow rate measurements were not possible at all sites because of inaccessibility of the tile line, extremely large flow rates, or both. Mass emission of nitrate-nitrogen was calculated for systems in which both concentration and effluent discharge rate were measured. The measured concentration and discharge rate at a given date were assumed to have remained constant for a period half-way to the previous and following monitoring dates. The calculated amounts for each monitoring period were then totaled to cover one year.

Data on fertilizer and water applications were collected from growers on sites monitored in the free drainage and tile drainage investigations.

The research conducted at the Davis plots contrasts the free drainage and tile drainage investigations. In the latter two cases, measurements were made on commercial operations under a variety of conditions.

There were no experimentally controlled management variables. On the other hand, the Davis plot investigation represents carefully managed, highly instrumented and replicated experimental plots. Three irrigation treatments were imposed during the growing season of a corn crop. These were application of water to replace 1/3, 3/3, and 5/3 of the water lost through evapotranspiration. These three levels of irrigation corresponded to approximately 20, 60, and 100 surface centimeters of water during the growing season. In addition to the irrigation treatments, there were four fertilizer treatments of 0, 90, 180, and 360 kg nitrogen/ha applied at the time of seeding. Estimates of nitrogen and water movement past the 300-cm soil depth during the 1974 and 1975 growing seasons are reported here. (The experiment and the analysis of the data have utilized more sophisticated methods of data collection and evaluation of mean values than most field experiments, particularly where rate processes are involved. Spatial variations of both the water flux and nitrate have been considered. A detailed description and analysis of the data will appear in other publications).

RESULTS AND DISCUSSION

As previously stated, the only data we presently have available on water management is the amount of water leached beyond the root zone for a given period of time. Therefore, we will be analyzing the relationship between nitrate-nitrogen leached and the drainage volume. Inasmuch as the nitrate available for leaching is expected to be related to fertilizer nitrogen application, we will also evaluate the relationship between nitrate leaching and fertilizer nitrogen input. Both mass emission and concentration of the leachate will be used as parameters for nitrate leaching. The following symbols will be used hereafter: C represents the nitrate-nitrogen concentration in mg/l; M is mass emission in kg/ha/yr; W represents the drainage volume under free drainage systems and the effluent volume in tile drainage systems and is expressed as ha cm/ha; and N is the fertilizer nitrogen applied in kg/ha/yr.

The amount of nitrate-nitrogen leached is plotted as a function of drainage volume in Figure 1 for sites studied under the free drainage investigation. Linear regression analysis on those data results in the equation $M = 11.7 + 3.05 W$ with $r = .766$*** (*, **, and *** will be used to denote statistical significance at the 5, 1, and 0.1% levels, respectively). The data plotted in Figure 2 show that there is no significant relationship between the nitrate-nitrogen concentration and drainage volume. The resultant equations and correlation coefficients obtained from linear regression analysis between various factors (for free drainage sites) are presented in Table 1. Mass emission had correlation coefficients significant at the 0.1% level when related to drainage volume, fertilizer input, or the product of the two variables. (Note that the highest correlation coefficient was achieved for mass emissions versus the product of drainage volume and fertilizer nitrogen application).

Table 1. Equations and correlation coefficients for nitrate-nitrogen leached and various factors for sites analyzed in the free drainage investigation.

Equation	Correlation coefficient	F
$M = 11.7 + 3.05 W$	.766***	79.4 ***
$M = 13.0 + 0.469 N$	.683 ***	26.3 ***
$M = 54.5 + 0.0067 NW$	.794 ***	51.1 ***
$C = 36.5 - .162 W$	-.220 NS	1.5 NS
$C = 24.9 + .0242 N$	.216 NS	1.4 NS

There was no significant correlation between the leachate concentration and either drainage volume or fertilizer input.

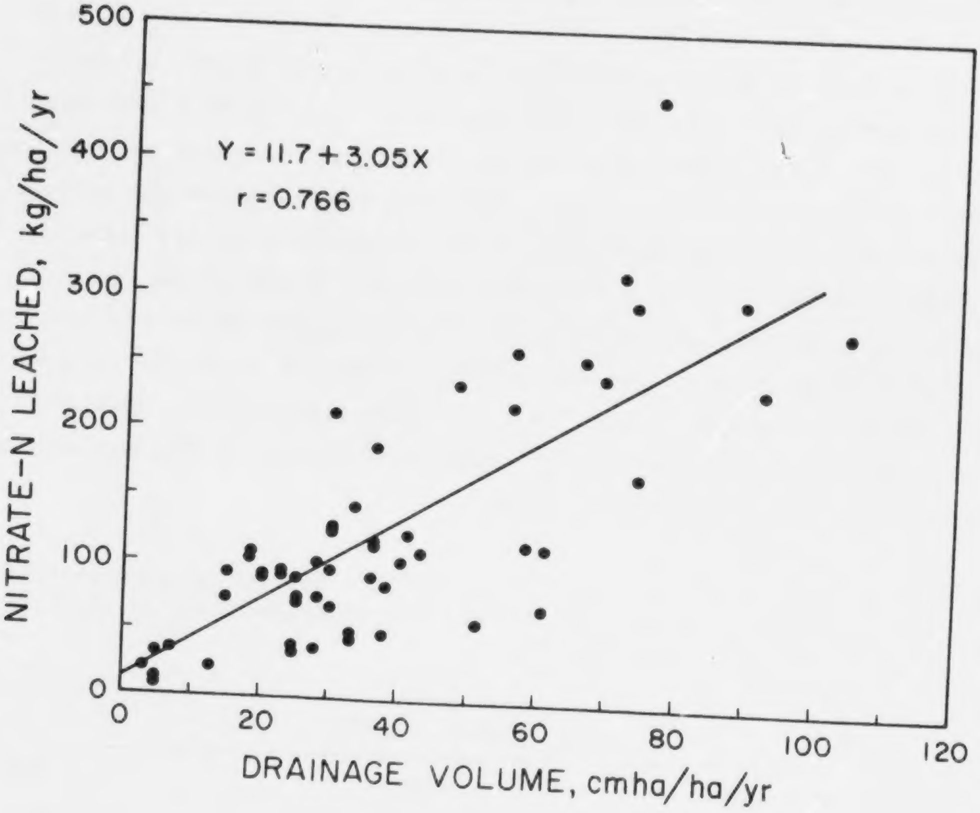

Figure 1. Correlation between the nitrate-nitrogen leached and drainage volume for 58 sites from coastal and interior valleys of California.

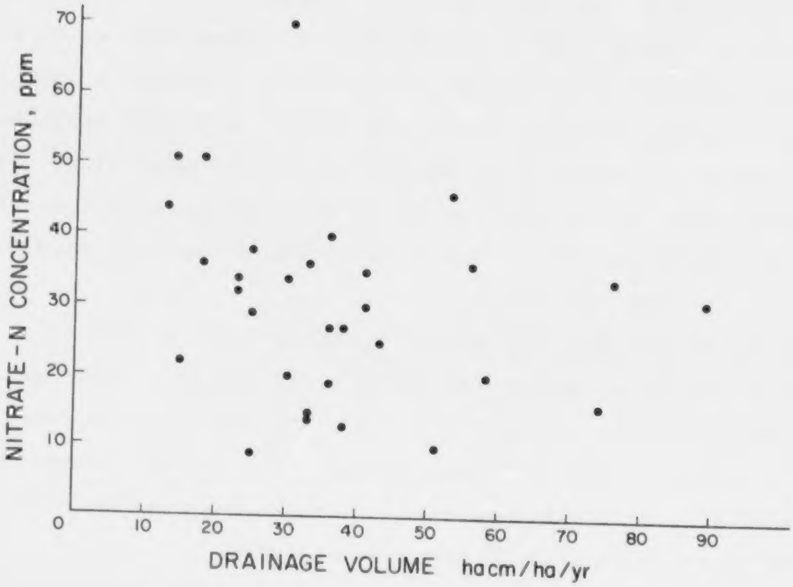

Figure 2. Relationship between concentration of nitrate-nitrogen in the drainage water and drainage volume for free drainage sites.

The amount of nitrate-nitrogen discharged during the given year from various tile drain systems is plotted in Figure 3 as a function of the amount of water discharged. Sites located on the Oxalis and Kettleman soil series on the west side of the San Joaquin Valley are suspected of having considerable amounts of native nitrogen in the soil profile. Data from those sites are identified in Figure 3 by open circles. Note that in general a higher mass emission of nitrate-nitrogen occurred for a given tile effluent volume for those systems as compared to other sites analyzed. Those sites which had alfalfa as a crop and no fertilizer input for several years are also separately identified in Figure 3 with open triangles. Note that as expected there was less mass emission per unit of tile effluent volume as compared to systems receiving fertilizer input. A linear regression analysis shown in Figure 3 is for all sites exclusive of the sites on the west side of the San Joaquin Valley and those having alfalfa as a crop. Note that there was an increase in mass emission with increased tile effluent volume.

The equations and correlation coefficients obtained from linear regression analysis for data accumulated in the tile drainage investigation are presented in Table 2. The statistical analyses were conducted using data from all sites and also for data accumulated from sites exclusive of those suspected of having high native nitrogen or very low nitrogen because of the alfalfa crop.

The highest correlation coefficient was obtained between mass emission and the product of effluent volume and fertilizer nitrogen application. Correlation coefficients significant at the 0.1% level were also obtained between mass emission and tile effluent volume for both sets of data. There was a reasonably high correlation coefficient between mass emission and fertilizer nitrogen application particularly when the sites suspected of having high native nitrogen and/or zero nitrogen input were excluded from the analysis. There was either no significant correlation or poor correlation between tile effluent

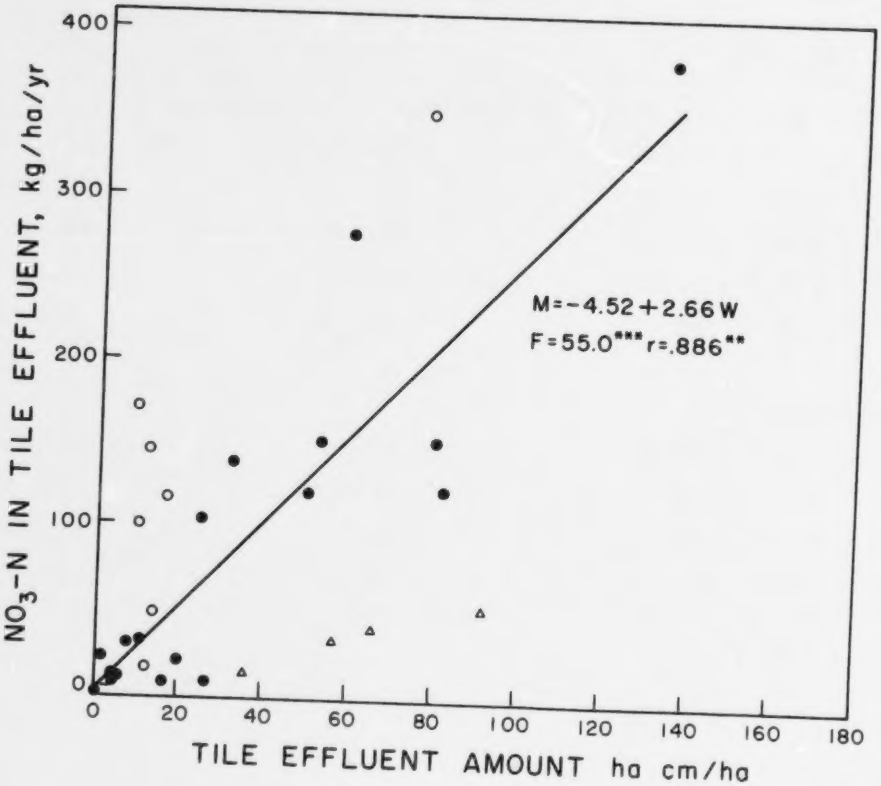

Figure 3. Relationship between mass emission of nitrate-nitrogen and effluent volume for tile drained fields. The open circles represent sites that had native geological nitrate and the triangles represent sites that were cropped with alfalfa and received no nitrogen fertilizer.

Table 2. Equations and correlation coefficients for nitrate-nitrogen leached and various factors for sites analyzed in the tile drainage investigation.

Equation	Correlation coefficient	F
Sites exclusive of alfalfa and high native nitrogen		
$M = -4.52 + 2.66\ W$	.886 ***	55.0 ***
$M = -48.9 + 3.82\ N$	.715 **	16.8 **
$M = 16.4 + .0042\ NW$	.920 ***	82.9 ***
$C = 7.92 + 0.36\ N$	.532 *	5.9 *
$C = 17.6 + .119\ W$	.290 NS	1.4 NS
All sites		
$M = 7.72 + 2.76\ W$	.840 ***	47.8 ***
$M = 66.1 + .4\ N$	.46 *	5.2 *
$C = 44.9 - .182\ W$	.14 NS	0.4 NS
$C = 29.4 - .0007\ N$	.006 NS	.001 NS

concentration and fertilizer input or tile effluent volume.

Significantly, results from the free drainage and tile drainage investigations produced similar types of results. In both cases, there was essentially no correlation between nitrate-nitrogen concentration and fertilizer input or drainage volume. The best correlation was found between mass emission and the product of fertilizer input and the drainage volume. The second best correlation was between mass emission and drainage volume and this was followed in significance by the relationship between mass emission and fertilizer input.

Clearly, both fertilizer input and drainage volume significantly affected the amount of nitrate leached below the root zone. A combination of both factors provided the highest correlation coefficient. Considering the factors independently, the amount of nitrate leached was more closely related to the amount of drainage than to the fertilizer input.

We previously discussed whether mass emission or concentration is the more appropriate parameter to represent nitrate leaching. We concluded that mass emission is the better parameter because it clearly represents the amount of a resource lost and serves as an indicator on the magnitude of the nitrate assault on the environment. Our data suggest that mass emissions can be reduced by reducing the drainage volume, fertilizer input, or both. Theoretically, at least, mass emissions can be managed. On the other hand, nitrate-nitrogen concentrations were not correlated with management variables. It must be recognized, however, that the correlations were carried out between two variables with all other variables being uncontrolled. For example, one would theoretically expect a higher concentration in the leachate with increased fertilizer nitrogen input if all other variables were constant. The lack of correlation between concentration and nitrogen input simply means that in the systems investigated other variables are acting on the system to remove the significant correlation. One would also expect lower nitrate-nitrogen concentrations with increased drainage volumes simply because of the dilution factor. The fact that this was not observed again indicates that there were other interacting variables in the system.

Water management resulting in high leaching volumes removes considerable nitrogen from the root zone. A grower may learn from experience that he needs to apply higher amounts of nitrogen (to compensate for the leached nitrogen) under his management system to have maximum crop production. Inasmuch as growers are not generally aware of the amount of their drainage volume or nitrogen leaching, their fertilizer practices would have been developed by experience. Our data allowed us to determine if there was a significant relationship between the drainage volume and fertilizer application. A linear regression analysis resulted in the equation $N = 78.8 + 4.07 W$, with $r = 0.618$ *** for the free drainage sites and the equation $N = 275 + 2.85 W$, with $r = 0.524$ * for the tile drainage sites. Therefore, we observed that fertilizer nitrogen application, in general, is indeed higher under those systems with higher drainage volumes. These results clearly demonstrate that fertilizer management and water management are inter-linked and one cannot be evaluated separately from the other.

We will now consider results from the Davis plots. These data were collected under carefully controlled conditions with detailed analytical analysis. The experimental approach on the Davis plots is in extreme contrast to the free drainage and tile drainage investigations. Gross analysis on systems with many uncontrolled and unmeasured variables were carried out on the latter two systems whereas sufficient precision to allow analysis of spatial variability in a "generally recognized uniform soil" was carried out in the former. Even though a given set of plots was irrigated the same, the calculated amount of water passing the 300-cm layer during the growing period was analyzed on different plots. For plots which were considered to be uniform, receiving the same treatment, each had unique drainage volumes and mass emissions. A linear regression analysis was conducted between mass emission and drainage volume from data collected in each plot receiving a given fertilizer treatment. The linear regression curves with their equations and correlation coefficients are illustrated in Figure 4. Note that r is significant to at least the 1% level for all fertilizer treatments. The highest value of r resulted from plots receiving 360 kg/ha of nitrogen. An intermediate value of r occurred for the 180 kg/ha fertilizer application and the lowest r values were obtained for the 0 and 90 fertilizer treatments. The slopes of the curves also appear to differ for different fertilizer treatments. The result of a statistical comparison of these slopes is presented in Table 3.

Table 3. The F values from the statistical comparison of the slopes of mass emission versus drainage volume curves and adjusted means for different fertilizer treatments at Davis plots.

Fertilizer application comparison	F slope	F Adjusted Mean
0 vs 90	.36 NS	2.72 NS
0 vs 180	1.51 NS	8.41 **
0 vs 360	18.71 ***	16.10 ***
90 vs 180	3.81 NS	2.91 NS
90 vs 360	25.06 ***	11.02 **
180 vs 360	10.70 **	5.47 *

A comparison between the adjusted means for different fertilizer treatments is also presented in Table 3. The 360 treatment is significantly different from all other treatments, both with respect to slope and adjusted mean. The slopes of the other fertilizer treatments are not significantly different from each other. The adjusted mean between 0 and 90 and also 90 and 180 are not significantly different but the

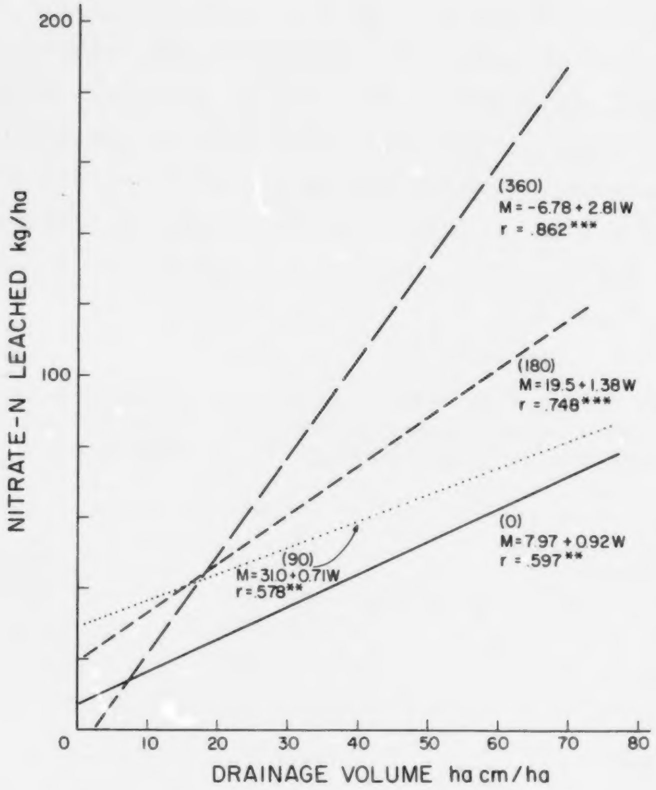

Figure 4. Relationships between nitrate-nitrogen leached and drainage volume for four different rates of fertilizer nitrogen application. The 0, 90, 180, and 360 numbers in parentheses refer to rates of nitrogen in kg/ha/yr.

adjusted means between 0 and 180 are significantly different at the 1% level. In making these comparisons, it should be recognized that the inherent fertility of all plots was high at the beginning of the experiment and that differences between fertilizer treatments would be expected to be magnified in successive years.

The results from the Davis plots provide significant evidence that mass emission is related to drainage volume. As might be expected, the effect of increasing drainage volume on mass emissions was higher at the highest fertilizer input.

Data from these three investigations suggest that mass emission is more strongly related to drainage volume than to fertilizer application. Indeed, we can note from Figure 4 that there was relatively little difference in the amount of fertilizer leached for the different application rates when the drainage volume was low. It was only at the high drainage volumes that the effects of different fertilizer treatments became evident. Conceivably, high fertilizer applications can be made without excessive pollution potential if the leaching volume is kept very low. On the other hand, if the drainage volume is high, considerable nitrate leaching can occur, even with low fertilizer application. Note in Figure 3 that increased drainage volume produced increased mass emission, even under systems receiving no nitrogen fertilizer in which alfalfa was the crop. Alfalfa is recognized as being a good nitrogen scavenger.

A comparison between mass emission and drainage volume for the free drainage, tile drainage, and the 360 kg/ha fertilizer treatment at Davis plots is made in Figure 5. The slopes of all 3 curves are very similar. Indeed, the tile drainage and the Davis plot curves are almost identical. These results are probably coincidental, but nevertheless they could lead to interesting speculations.

In conclusion, the results of all three studies clearly indicate that nitrate leaching is significantly affected by water management.

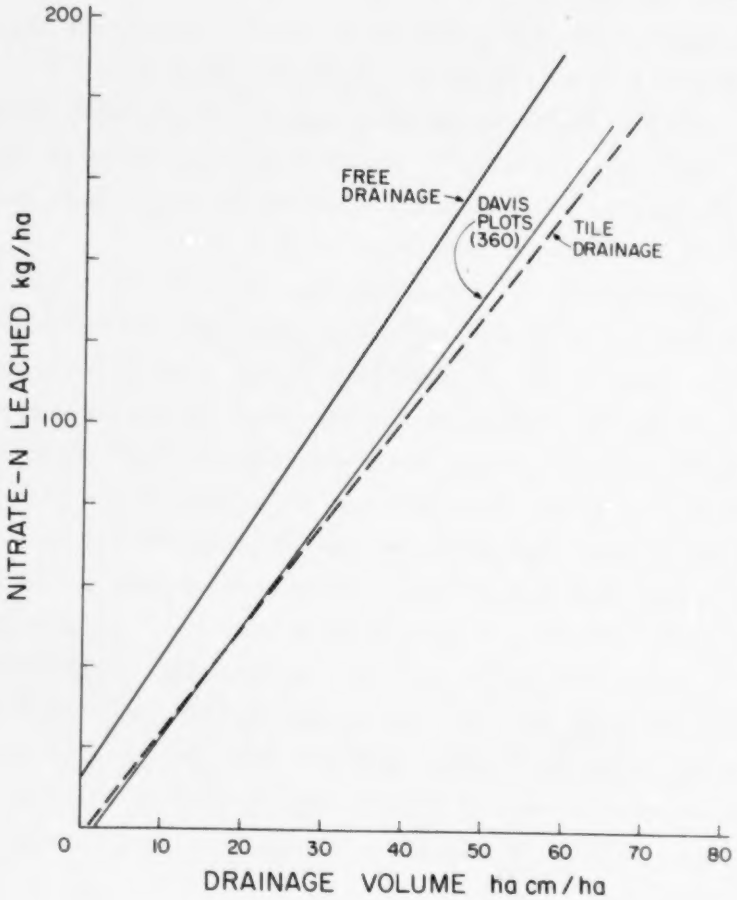

Figure 5. Relationship between mass emission and drainage volume for free drainage and tile drainage sites and for the 360 kg/ha nitrogen rate on the Davis plots.

Management which results in very low drainage volumes contribute relatively low mass emissions of nitrate. On the other hand, water management resulting in high leaching volumes will cause considerable leaching of nitrates. Water management appears to have very little effect on the concentration of nitrate-nitrogen beyond the root zone, presumably because fertilizer management has been adjusted to compensate for the higher leaching.

A strategy for reducing nitrate-nitrogen leaching would be to reduce the drainage volume. This alternative, however, may require investment in irrigation systems which allow more precise control of water application. Furthermore, added instrumentation may be required to guide irrigation practices to produce minimum leaching. In some cases, there may be a trade-off between low nitrate-nitrogen leaching with increased capital and energy investment.

MONITORING WATER FOR NITROGEN LOSSES FROM CROPLANDS

Kenneth K. Tanji[1]

ABSTRACT

This paper reviews monitoring for nitrogen in surface and subsurface return flows from irrigated lands.

Presented are the elements of a monitoring program: objectives of monitoring, parameters to be measured, sampling programs that include site selection, sampling frequency and sampling method, laboratory methods for nitrogen determinations, requirements for resources and facilities, evaluations of collected data and other supporting information and data, and dissemination of monitored results.

Each of these elements are appraised and a conclusion is drawn on monitoring waters for nitrogen losses from croplands.

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INTRODUCTION

This paper focuses on the questions of why, what and how to monitor nitrogen emissions in surface and subsurface return flows from irrigated agriculture. How much nitrogen is likely to be leached from soils, lost to the atmosphere in gaseous forms, and taken up by crop plants are addressed by other papers in this volume. This report compliments other conference papers dealing with sources and forms of nitrogen in soils and waters, as well as basin-wide field investigations and modeling.

It should be mentioned that the State of California is implementing a statewide water quality surveillance and monitoring program as mandated by the California Porter-Cologne Water Quality Control Act of 1970 and to satisfy the requirements of the Federal Water Pollution Control Act Amendments of 1972 (PL 92-500). For the National Pollutant Discharge Elimination System (NPDES), permittees from irrigated agriculture are required to report the water flow, electrical conductivity, and suspended solids in their supply and discharge waters. The State Water Resources Control Board (SWRCB) does not have a formal statewide monitoring requirement for nitrogen emissions from croplands. However, where nitrogen is a problem, it is being monitored in the receiving water and return flows by federal, state, or local entities. It is not certain what impact(s), if any, the recent Federal Water Pollution Control Act Amendments of 1978 (PL 95-217) will have on irrigated agriculture other than that collected irrigation return flows are now reclassified as non-point sources.

Recognizing that nitrogen in return flows from irrigated agriculture is monitored only in localized problem areas in California, an attempt is made herein to describe the elements of a possible monitoring program. Presented are some details on the objectives of monitoring, identification of what needs to be measured and how to measure them, and what to do with this monitored data.

ELEMENTS OF A MONITORING PROGRAM

A monitoring program should include the following elements:

1. Objectives of monitoring.
2. What parameters to measure.
3. How and where to sample in the system.
4. Which laboratory analytical method to use.
5. Requirements for resources and facilities.
6. Evaluation of monitored data.
7. Dissemination of monitored results to information users.

In the following sections, each of the above elements will be discussed relative to monitoring nitrogen emissions from croplands in surface and subsurface return flows.

OBJECTIVES OF MONITORING

The aim of monitoring nitrogen losses from croplands, ideally, is to satisfy a number of information and data needs. The objectives may include:

1. Obtain baseline data to define existing nitrogen status.
2. Detect developing problems at the earliest stages.
3. Evaluate nitrogen fertilizer and water management practices.
4. Appraise land disposal of nitrogen-containing wastewaters and wastes.
5. Evaluate potential reuse of return flow waters and/or impacts on the beneficial uses of receiving waters.
6. Acquire input data as well as test data for models that examine cause and effect relations, and management alternatives and anticipated changes.
7. Provide guidelines for water quality management plans, i.e., 208 Planning.

As noted above, the purposes of monitoring could be diverse and there is some question on whether any one selected monitoring system would be able to satisfy all or most of these objectives.

IDENTIFICATION OF MEASUREMENT PARAMETERS

Nitrogen

The various forms of dissolved and particulate (suspended) nitrogen that may be present in surface and subsurface irrigation return flows are:

1. Dissolved nitrogen gases (dinitrogen, ammonia, etc.)
2. Soluble inorganic nitrogen (ammonium, nitrate and nitrite)
3. Soluble organic nitrogen (amino acids, sugars, etc.)
4. Particulate organic nitrogen (suspended matter consisting of plant and animal origins).
5. Sorbed inorganic nitrogen (exchangeable and fixed ammonium in sediments, etc.).

The discharge of the above forms of nitrogen varies with site-specific conditions. Items 1-5 may be present in surface runoffs and items 1-3 in percolating and subsurface drainage waters. The nitrogen species that are most frequently measured in supply and return flows are nitrate or nitrate plus nitrite, less frequently ammonium and ammonia, and infrequently dissolved and suspended organic nitrogen.

Ammonium, and ammonia in high pH waters, should be more frequently analyzed because it is an indicator of pollution, and of inefficient use of ammonia/ammonium fertilizers, and is highly toxic to aquatic organisms. Nitrate is of importance because it plays a role in eutrophication of surface waters, in high concentrations may cause methemoglobinemia, may be indicative of excessive leaching, and may be detrimental to certain crops during the maturation stage. Significant concentrations of nitrite may occur only under unusual anoxic (reduced) conditions. Organic nitrogen present as nitrogenous fraction of biochemical oxygen demand (BOD), when BOD determinations are made for more than 5 days of incubation, is also indicative of pollution, and may cause a lowering in dissolved oxygen (DO) in the receiving waters as well as contributing mineralized forms of nitrogen to waters.

In surface runoff from close-growing crops, such as pasture and flooded rice fields, organic nitrogen usually dominates over the mineralized forms while in widely-spaced crops, such as furrow irrigated field crops, ammonium and nitrate may be more prominent than organic nitrogen.

Other Significant Parameters

Along with the measurement of nitrogen species in surface and subsurface return flows, water flow should be estimated so that both the concentration and mass emission (concentration times volume) of nitrogen can be considered.

Since the concentration and mass of nitrogen in return flows are affected by site-specific conditions and local management practices, it is necessary to obtain additional information and data to more fully evaluate nitrogen losses from croplands. The following are some of the principal factors that influence the quantity and quality of irrigation return flows:

Quantity - availability and cost of supply water, irrigation application methods and attainable efficiencies, special cultural practices, extent of water reuse at the on-farm, district and basin levels, constraints on water reuse due to the presence of excess salts, boron, sodium or chloride, and the response or lag-time in water flows.

Quality - supply water, presence of nitrogen native to the soils, nitrogen fertilizer practices and management, amount of nitrogen taken up by crops, discharges into irrigation drains by other sectors of society and land uses, and disposal of animal wastes and wastewaters in croplands.

SAMPLING PROGRAM

How and where to sample for nitrogen in surface and subsurface return flows from croplands are difficult to answer. Ideally, the sampling should represent the system being monitored in both time and space. It begs the questions of sampling site selection, sampling method, and sampling frequency.

Site Selection

The main consideration for where to sample is the identification of the principal water flow pathways and related physical and operational characteristics of the system being monitored. In the case of irrigated croplands, the main components of return flows are surface runoff (irrigation tailwater, precipitation runoff, operational spills from distribution systems), collected subsurface drainage (effluents from tile drainage systems and drainage wells, subsurface waters intercepted by natural and man-made open channels), and percolating drainage water. With some exceptions, the surface runoffs and collected subsurface drainage waters are not handled separately in the drainage network, and the combined flows from these two major components are referred to as collected surface irrigation return flows.

At the systems level (for instance, irrigation district) of monitoring, sampling stations are usually located at the outermost boundaries where surface return flows from the given system discharge into a contiguous drainage area or receiving water body. In addition, for such large tracts of croplands, it is desirable to establish internal sampling stations in the drainage network to ascertain subsystem (local) differences. At the on-farm and field level of monitoring, it may be desirable to separately monitor the various components of collected subsurface drainage and surface runoff waters.

Those components of return flows that are collected and/or controlled are more amendable for estimating water flows and sampling for nitrogen than those that are uncollected (diffuse) and/or uncontrolled. In some instances, however, estimating surface water flows can become quite complex when the direction of flow of water changes. For uncollected, uncontrolled flows such as water percolating beyond the crop root zone, it is extremely difficult to measure them directly, and perhaps should be obtained by difference.

Other criteria for site selection may be monitoring objective, ease of accessibility to the sampling site and available resources.

Sampling Frequency

The frequency of sampling return flows should be dictated by the system behavior and the nature of the data characteristics representing this behavior. A grab sample gives instantaneous characteristics for a given time and a given place. A composite sample over a finite time interval gives the average characteristic for that time interval. Sequential and continuous samplings usually offer a better opportunity of obtaining patterns of fluctuations and trends.

When large changes in water flow or nitrogen concentrations are expected over a short time period, the frequency of sampling should be increased, and, conversely, when these changes are small, the frequency of sampling may be lengthened. At any rate, it is desirable to have both time-integrated measurements over a fixed point in space as well as space-integrated measurements over a fixed time.

Sampling Method

Although continuous water sampling and water flow measurements may be desirable, this is frequently not possible for all the components of return flows because of costs and/or lack of suitable sampling and gaging equipment.

Continuous surface water flow measurements can be obtained by a variety of methods, but most frequently it is measured with a stilling well and waterstage recorder for open channels and a volumetric water meter for a pipeline. Instantaneous surface water flows are estimated by measuring head on a weir or by a current velocity meter if there are no structures. For tile drainage effluents discharged by gravity flow into an open channel, instantaneous flow estimates may be obtained by the trajectory method or the bucket and stop watch method. Tile drainage effluents collected in a tile sump and pumped into an open drain are more amendable to flow measurements by knowing the electrical charges and pump efficiency. This method of flow estimate is also applicable to drainage wells. As mentioned previously, direct measurements of flow of percolating waters are usually not possible, at least on a routine basis.

As for sampling water for nitrogen, automated portable sequential and composite samplers are available. A composite sampler collects a single water sample by setting a fixed time interval between sampling while a sequential sampler collects separate water samples according to a preset time interval and sample volume. Some automated samplers are equipped with flow meters so that they can operate on intervals of flow pulses. Since nitrogen species are subject to biological transformations provisions for chilling with ice or refrigeration must be made during the collection, transportation, and other handling stages up to the point of chemical analysis.

Water samples are also obtained by taking periodic grab samples. If particulate forms of nitrogen are being analyzed, the usual precautions and techniques in sampling that apply to sediment sampling should be followed, especially in fast-flowing open channels. Water may be also obtained from observation (piezometric) wells to sample the water table and from wells to obtain the "pumped" groundwater.

Nitrogen in the soil is generally sampled in two ways. One method samples the soil profile with hand- or machine-operated soil augers followed by measurement of the water-soluble nitrogen in the soil at some reference laboratory soil water content. The other method involves placement of a porous ceramic cup (suction probe) in the soil profile and extraction by vacuum. Soil solutions, however, can be obtained only up to a vacuum of about 20 inches of mercury, i.e., only when the soil is moist.

LABORATORY ANALYTICAL METHODS

Approved analytical methods for various nitrogen species are found in a number of references such as Standard Methods for the Examination of Water and Wastewaters (APHA, AWWA, WPCF), American Society for Testing and Materials (ASTM), Methods for Chemical Analyses of Waters and Wastes (EPA), Methods for Collection and Analysis of Water Samples for Dissolved Minerals and Gases (USGS), and Methods of Soil Analysis (ASA).

Although other analytical methods and procedures not approved by the organizations above are available, their accuracy and precision may be of some concern. Even with the use of approved methods, some quality control should be exercised because of potential variabilities in the use of differing chemical reagents and instruments, performance and capabilities of the analysts and sample presentation, preparation and handling.

RESOURCES AND FACILITIES

The extensive and intensive aspects of the monitoring program is influenced strongly by available resources and facilities, which includes wages and salaries, transportation and travel expenses, analytical costs, purchases and maintenance of equipment and machinery, etc. Trade-offs are frequently made between the scientific-engineering requirements for monitoring and what is economically feasible. The surveillance type of monitoring is generally less costly than those which attempt to identify and control the sources of nitrogen discharged from croplands.

The bottom line, for the frequency of sampling, number of sampling sites, and the methods of sampling are the associated costs. In spite of all the desirable sampling features and requirements that were pointed out, monitoring nitrogen in return flows usually reduces to a limited number of sampling stations, a limited sampling frequency, and a limited number of nitrogen species analyzed.

DATA EVALUATION

The most difficult and perhaps the most important task in the monitoring program is adequate evaluation of monitored information and data.

The monitored data is usually characterized by minimum and maximum values as well as arithmetic means. The arithmetic mean concentration of nitrogen in waters should preferably be the flow-weighted average rather than a simple average of the concentrations obtained. In addition, seasonal trends such as irrigation versus nonirrigation seasons and other data fluctuations are reported in tabular or graphical forms as a function of time and/or space (distance).

Since errors may be introduced in the sampling program as well as the analytical determination for nitrogen, the monitored data should be subjected to further statistical analyses such as standard deviation and standard error. The latter error can be controlled to some extent by quality control but the former is more difficult to appraise and control. The question of whether the sampling program is a valid sampling of the population is of concern here.

Some parameters exhibit normal (Gaussian) distribution while others may be log-normal, bi-modal, or have some other distribution characteristics. If the population is not normally distributed, the statistical analyses become more complex and sampling must be more extensive. Currently, the spatial variability of the leaching characteristics of field soils is receiving much attention where spatial variability is pronounced, substantial errors can be made in point-type measurements if the mass of nitrogen is taken as the product of average nitrogen concentration and average water flow. To minimize this kind of error, the number of samplings over both time and space must be significantly increased. In other space-integrated or time-integrated types of measurements, this problem may be minimized. For instance, a tile sump collecting tile drainage effluents from a field usually would yield better measurements for the concentrations of nitrogen percolating to the water table than the use of suction probes which is a point sampling method.

Besides the nitrogen and water data obtained in the sampling program, other supporting information and data base are required to more fully appraise the results of the monitoring program. These supporting information and data are the previously mentioned factors influencing the quantity and quality of irrigation return flows. Such information and data would increase the utility of the monitored data and may help in ascertaining cause and effect relationships.

DISSEMINATION OF MONITORED RESULTS

Too often the information and data collected are used solely by the party responsible for the monitoring program. Although many do publish water quality and water flow data or submit such data into data storage and retrieval banks like the STORET System, this does not fully satisfy the objectives of monitoring nitrogen in irrigation return flows. Some exceptions are the detailed reports that contain not only the monitored data but also supporting information and data mentioned previously along with some interpretation.

CONCLUSIONS

Monitoring for nitrogen in surface and subsurface return flow waters from croplands is easier said than done. Although the goals of monitoring nitrogen emissions may be definitive, the design, implementation, operation and data evaluation of present day monitoring efforts generally fall far short of these goals. As Professor D. R. Nielsen of the University of California at Davis has stated: "We are still learning how to monitor the out-of-doors". That is not to say that we should cease to monitor for nitrogen from croplands, but to be cognizant of the deficiencies and strive to make some improvements.

From a water quality and pollution control point of view, as well as from a point of view of agricultural production, this author suggests that emphasis be placed upon developing and applying best management practices along with some limited effluent monitoring dictated by the degree of actual or potential problem.

ACKNOWLEDGEMENT

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ESTIMATING THE INFLUENCE OF SOIL RESIDENCE TIME ON EFFLUENT WATER QUALITY

W. A. Jury[1]

ABSTRACT

Models are proposed to calculate the time required for dissolved chemicals to move from the soil surface to either underlying groundwater in the case of free drainage or to tile drain outlets in the case of artificial drainage. The models assume that dissolved substances are transported primarily by moving soil solution, which displaces soil water initially present in the wetted pore space (piston flow approximation). For free drainage, this results in a single travel time equation which is a function of soil water content and drainage volume. For tile drainage, the travel time depends also on the surface entry point, which is illustrated with a graph showing residence time as a function of place of origin. This graph may be used for all tile drain systems.

Calculations are presented to show the influence of travel time on drainage concentrations, illustrating also how tile drain concentrations are a mixture of contributions from different parts of the field arriving at different times.

Field studies of salt movement are analyzed and compared to the model predictions, with differences explained on the basis of soil variability or stagnant water in part of the wetted pore space.

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INTRODUCTION

Concern has risen in recent years over the contribution of agriculture to rising nitrate concentrations in streams and groundwater (Ayers and Branson, 1973; Bingham et al, 1971). As a result, a number of investigators have been measuring nitrate concentrations in drainage water below crop root zones and at tile drain outlets. As numerous researchers have pointed out, however, nitrate concentrations below the root zone alone will not give a good estimate of the pollution potential of a field, but must be coupled with an estimate of the drainage volume in order to calculate the amount of nitrate moving downward (Letey et al, 1977). For artificially drained systems, the amount of nitrate leaving a field may be calculated from a simultaneous measurement of concentration and tile discharge, but for free drainage systems water flow below the root zone is more difficult to measure.

An important characteristic of nitrate movement in soil is the time required to reach the groundwater or tile drain from the soil surface. Knowledge of this travel time is important for determining the impact of a change in surface practices on groundwater concentrations and for identifying the source of a pollutant detected in the soil.

Determination of travel time is difficult in the field for a number of reasons. In free drainage, water flow rates below the root zone are hard to measure or even to estimate, and vary from point to point under the field (Biggar and Nielsen, 1976). In tile drained systems, the water flow path to the drain depends on where the water first entered the soil, so that chemicals introduced at the surface and moving with the soil water will require different lengths of time to reach the drain. As a result, the tile outlet concentration is a complicated space and time average of soil water concentrations within the field.

This paper will discuss ways of calculating and measuring travel time for nitrates moving downward to the groundwater (free drainage) or moving to a drain tile or ditch (artificial drainage). Research studies on this topic will be reviewed and summarized. Problems encountered due to variability in soil physical characteristics in the field will be discussed.

REVIEW OF RELATED RESEARCH

Free Drainage

In the absence of nitrate reduction by denitrification or plant uptake, nitrate should move through the soil in a manner similar to the chloride ion. Conclusions about ion transport based on chloride studies, therefore, should be applicable to nitrate also.

Warrick et al (1971) studied the movement of chloride through a clay loam during infiltration and found that a chloride pulse injected with water at the soil surface moved somewhat faster than predicted by a model which assumed that the infiltrating solution moved through all the wetted soil pore space. They also observed that the chloride pulse spread out rapidly due to dispersion while moving through the top 150 cm. This finding was similar to that of Miller et al (1965) who concluded that ponding water on the soil surface was less effective in moving salt from the root zone than the same volume of water applied intermittently because the leaching solution tended to move preferentially through the largest wetted pore space.

Endelman et al (1974) studied leaching of solid fertilizer nitrate and chloride in Plainfield sand under daily water applications of 2.5 cm. Each addition caused nitrate and chloride movement of 15 to 20 cm/day, slightly less than would be predicted from a perfect displacement (piston flow) model. Shuford et al (1977), in a similar experiment in silt loam, found that a nitrate and chloride pulse applied in solution moved faster through the soil than piston flow would predict. However, when this model was applied to an identical experiment on a Hawaiian soil, nitrate movement was overpredicted by a factor of three (Balasubramanian et al, 1973).

Other workers have observed that water moved preferentially through cracks and channels in the soil, shortening the travel time for surface applied chemicals and decreasing leaching efficiency for chemicals in soil solution (Kissel et al, 1975; Tyler and Thomas, 1977).

Movement of nitrate varies significantly with water input at the soil surface. Saffigna et al (1977) found that nitrate had moved 400 cm to the water table within 11 months after application to an irrigated potato field. A lysimeter recorded 80 cm drainage past the 150-cm depth during this time. However, Sommerfeldt et al (1965) found that nitrate applied to dryland grass had moved only 180 cm in 8 years where the annual average precipitation was 41 cm.

A few investigators have measured nitrate movement at great depths. Pratt et al (1972) and Adriano et al (1972, 1973) measured nitrate concentrations to 15 or 30 m below irrigated fields and estimated travel time by a piston flow model. They calculated values from 7 to 50 years to travel 30 meters, depending on the surface management and soil characteristics.

Recent investigations have observed great variability in solute movement at different locations within a field. Biggar and Nielsen (1976) estimated that 100 observations would be required on a 150 ha field to estimate the mean water velocity within ±50% of its true value. Jury et al (1976) found nonuniform infiltration of water and nitrate under sprinkler irrigated potatoes with nitrate penetrating much faster under the furrows than under the shoulders of the hills where the potato tubers were located.

Tile Drainage

Travel time for the drained fields is more difficult to estimate because of the uneven contributions from different parts of the field. Much of the work on nitrate movement has consequently focused on relating input of nitrate to output concentrations or nitrate accumulation at the tile line.

Glandon (1971) and Letey et al (1977) monitored numerous tile outlets in California for nitrate concentration and accumulation and found that nitrate concentration was not well correlated with water input, nitrogen input, or nitrate output flow. Nitrate flux through the tile was correlated with water flux through the tile in both investigations.

The ratio of nitrogen drained to nitrogen applied during the same year is frequently quoted by investigators monitoring tile drains, but this index may be misleading without some knowledge of the travel time from surface to drain (Jury, 1975). Johnson et al (1965) monitored drainage from four cropped fields for one year and found a range from .09 to .7 in this ratio. Meek et al (1969) found only .05 for the ratio of (nitrate out/nitrate in) during 5 months of monitoring an irrigated field. Calvert and Phung (1973) observed a ratio of 0.35 after one year for a field in Florida.

Several reported studies looked at leaching efficiencies for various surface water applications. Sadler et al (1965) leached a small experimental plot of chloride by continuous ponding and dropped the chloride concentration to .1 of the initial value after 145 hours. Talsma (1967) leached field plots of chloride with continuous and intermittent ponding and found the latter more effective in removing salt located far from a drain.

Jury (1975) proposed a method for calculating the travel time for tile drained fields and showed that the predictions of his model were consistent with the published observations of tile drain research studies, but often lent a different physical interpretation to the data than the one offered by the authors. This model will be discussed in greater detail below.

MODELS FOR CALCULATING TRAVEL TIME

Nitrate Movement in Soil

Nitrate ions are transported through the soil by three mechanisms: i) bulk flow (convection) within the moving soil solution, ii) self-diffusion by molecular collision with other ions, iii) hydrodynamic dispersion by mixing processes caused by local variations in water flow velocity and water flow direction due to the presence of the porous medium. Transport by diffusion is usually small compared to convection, and hydrodynamic dispersion is important only in cases of high solution velocity or large inhomogeneities (i.e. cracks) in the soil profile. In all applications, the mechanisms of diffusion and dispersion are used to describe spreading and mixing of solute pulses, and need not be included in an approximate calculation of travel times for surface-applied nitrate pulses.

Using the approximation that nitrate transport is due to moving solution alone, we may write

$$J_N = C_N J_W \qquad (1)$$

where J_N is nitrate flux (gm/cm^2/day)

J_W is solution flux (cm^3 soln/cm^2/day)

C_N is nitrate solution concentration (gm/cm^3 soln).

PISTON FLOW MODEL

The piston flow model is used by many individuals to approximately describe how water and dissolved chemicals move through the soil. This model assumes that solution applied to soil will replace the soil water previously in the pores, effectively pushing it through the soil like a piston. Figure 1 will help to illustrate the concept.

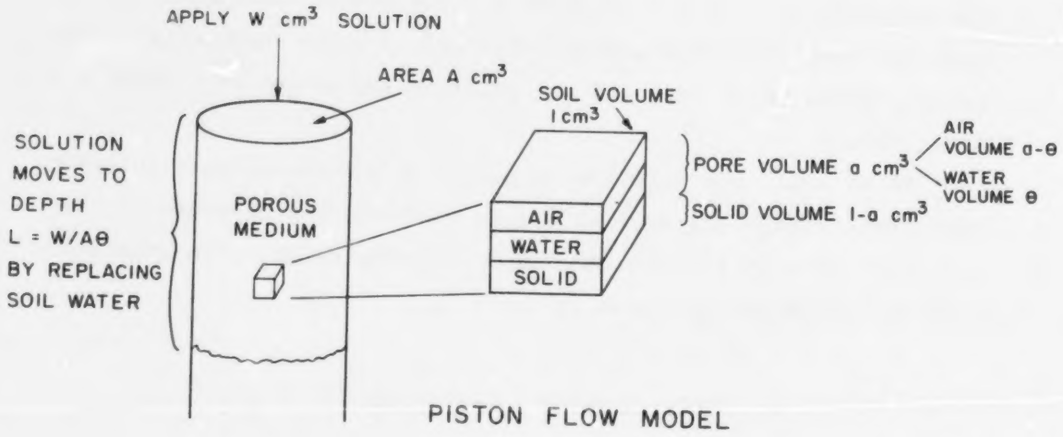
APPLY W cm³ SOLUTION
AREA A cm³
SOIL VOLUME 1 cm³
SOLUTION MOVES TO DEPTH L = W/Aθ BY REPLACING SOIL WATER
POROUS MEDIUM
AIR
WATER
SOLID
PORE VOLUME a cm³
AIR VOLUME a-θ
WATER VOLUME θ
SOLID VOLUME 1-a cm³
PISTON FLOW MODEL

Figure 1. Illustration of the piston flow model.

A long column of soil has a cross-sectional area A cm^2. Each volume element of soil contains a fraction 1-a of solid soil particles and a fraction a of water and air. The quantity a is called the fractional pore space or porosity. The fraction θ of the total volume filled with water is called the volumetric water content (cm^3 water/cm^3 soil).

If a volume W cm^3 of solution with concentration Co is fed into the top of this column (which we assume contains pure water C = 0), the solution will move to a depth L. The piston flow model assumes that the input solution pushed out W cm^3 of water initially in the column, which means that the new solution has moved to a depth $L = W/A\theta$ (cm).

If we apply the solution at a rate Q (cm^3/day), this solution will move through the soil at a rate $Q/A\theta$ (cm/day) because it can only move through the area $A\theta$ which contains water. The quantity $Q/A\theta$ is called the solution velocity V_W.

FREE DRAINAGE TRAVEL TIME

The time required for a pulse of nitrate to move a distance L (cm) through the soil in the piston flow approximation is simply

$$t = L/V_W \qquad (2)$$

where V_W is the solution velocity (cm/day), provided V_W is constant.

Movement Below Root Zone

Below the root zone, V_W may be approximated by

$$V_W = R/\theta \qquad (3)$$

where R is average drainage flux (cm^3/cm^2/day) and θ is average volumetric water content (cm^3/cm^3). If the water content in the root zone is changing then

$$R = I + P - ET$$

where I is the irrigation, P is precipitation applied at the soil surface and ET is evapotranspiration (consumptive use). Combining Eq (2) and (3) gives, for the time required to travel a distance L below the root zone

$$t = \frac{L\theta}{R} \tag{4}$$

The travel time for free drainage is thus seen to depend on surface management (I), external weather conditions (P, ET), and soil type (θ).

Movement Within Root Zone

The travel time for movement through the root zone is more complex, because the soil water flux J_W is not constant, but decreases from a value I + P at the surface to R at the bottom of the root zone. Furthermore, I + P is not constant with time and single irrigations or storms may leach nitrate through the entire root zone. Only in cases of frequent irrigation may we use a steady state model for water flow near the surface. Even in that case, however, Eq (2) must be replaced by a more complicated expression. Since J_W is not constant, the solution velocity $V_W = J_W/\theta$ is not constant either. One way of averaging V_W is to set $J_W = (I + R)/2$ so that the time required to travel through the root zone is

$$t = 2l\bar{\theta}/\ (I + R) \tag{5}$$

where l is the depth of the root zone and $\bar{\theta}$ is the average water content. Equation (5) will usually underestimate t.

Some simple examples will help to illustrate the use of Eq (4) and (5).

EXAMPLE 1

A quantity of nitrate near a sandy soil surface is moved downward by a series of rain storms totalling 20 cm. Assuming a water content of 0.15 (cm^3/cm^3) and negligible evaporation, calculate how deep the nitrate will be moved.

Since only 15% of the soil area is wetted, 20 cm rain will displace soil solution to 20/.15 = 133 cm.

EXAMPLE 2

A field under continuous cropping has an annual water application (irrigation + precipitation) of 140 (cm/year). The crop water use and soil evaporation are 80 (cm/year) and the depth of rooting is 1 (m). If the average water content is 0.25, calculate how long it will take nitrate to travel 25 (m) from the bottom of the root zone to the groundwater.

Drainage R is 140-80 = 60 cm/year; solution velocity V_W is 60/.25 = 240 cm/year; travel time is $t = R/V_W = 2500/240 = 10.4$ years. The approximate time to move to the bottom of the root zone by Eq (5) is 2 x 100 x .25/(60 + 140) = .25 year.

ARTIFICIAL DRAINAGE TRAVEL TIME

Determination of travel times for nitrates moving through ditch or tile-drained fields is more difficult than the free drainage calculation because the water flow path is curved from the surface to the tile or ditch line. For this reason, locations on the surface near the tile line will have much shorter flow paths and hence shorter travel times than locations near the midpoint between lines. To illustrate the procedure for determining travel time for artificially-drained fields, the following situation will be considered.

1) Uniformly spaced tile lines 2 S apart at depth d below surface
2) Steady water flow with uniform net water entry rate R

3) Impermeable barrier at depth d + D below surface

4) Isotropic soil properties

5) No crop in unsaturated zone

With this ideal system, shown in Figure 2, the movement of nitrate initially on the surface may be divided into two stages i) vertical flow to the water table, ii) curved flow (in the saturated region) converging to the drain line. The first stage is identical to free drainage and has a travel time

$$t_i = d\overline{\theta}/R \tag{7}$$

where $\overline{\theta}$ is the average water content between the surface and the water table.

The flow lines for the saturated region may be calculated from a mathematical model (Kirkham, 1958). A plot of these lines, called streamlines, is shown in Figure 3 for the region below the tiles in Figure 2.

The curved lines, representing the flow path for ions moving toward the drain, may be used to calculate the travel time by the following procedure. Two adjacent stream lines form a stream tube, shown in Figure 3. Since these lines are along the flow path, fluid will enter or leave the tube only at the ends, and the amount of solution required to move an ion through the tube is equal to the amount of fluid in the tube. The time required for this transit is the fluid volume in the tube divided by the entry flow rate.

Our model considers the tile lines to be very long and parallel, so that no changes in flow characteristics take place in a direction parallel to the drain. Our analysis will assume a unit thickness in this direction so that we will be working with area units rather than volume units. If the area of the stream tube in Figure 3 is a cm^2, then the amount of fluid inside is $\theta_S a$ where θ_S is saturated volumetric water content. The solution flow rate into the tube is

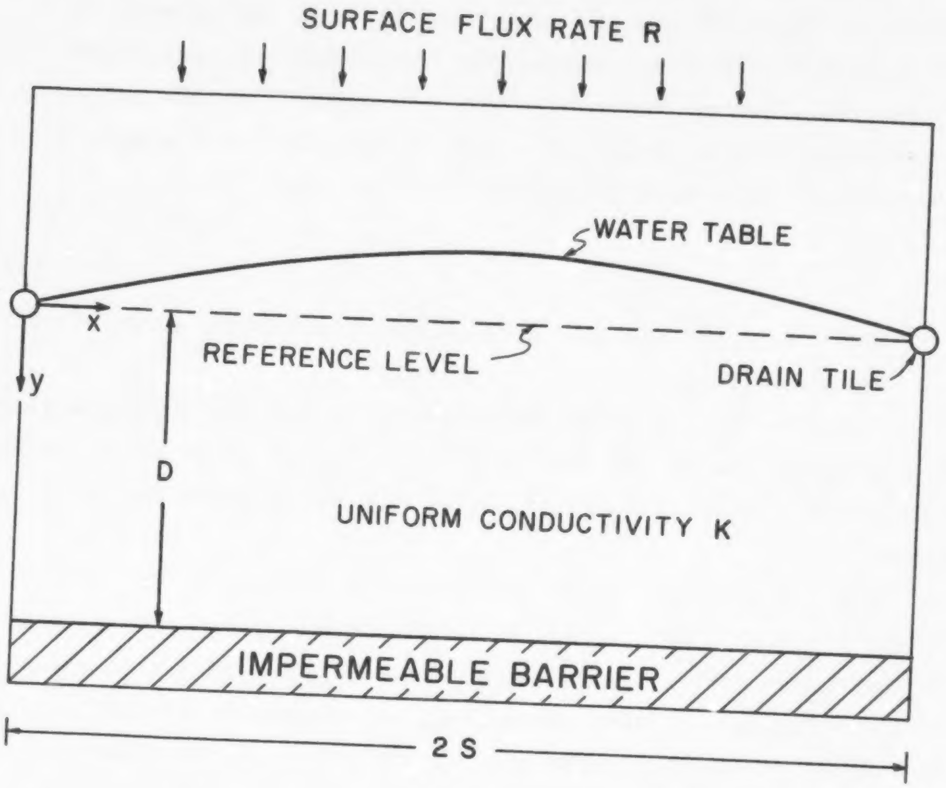
SURFACE FLUX RATE R
WATER TABLE
x
REFERENCE LEVEL
y
DRAIN TILE
D
UNIFORM CONDUCTIVITY K
IMPERMEABLE BARRIER
2 S

Figure 2. Ideal tile drained field with steady water uptake.

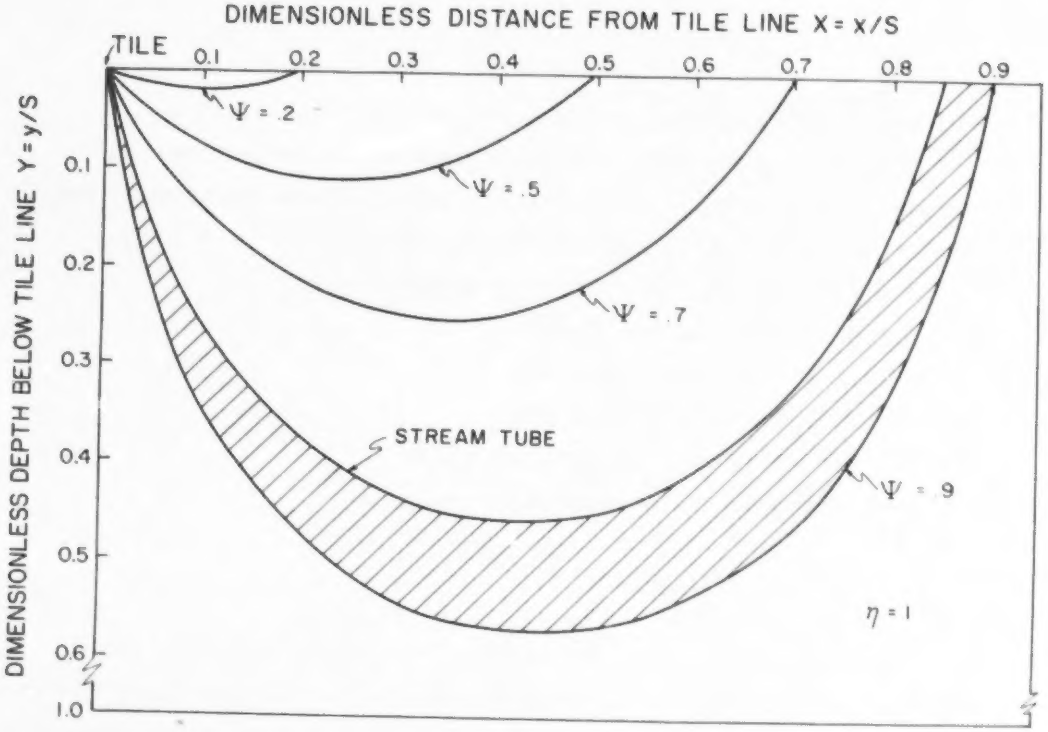

Figure 3. Streamlines for tile drained field.

$R\Delta x$ where Δx (cm) is the width of the tube at the entry point (water table) and R is the net drainage (cm/day). Thus, the travel time to go from the water table to the drain is

$$t = \frac{\theta_S a(x)}{R\Delta x} \tag{8}$$

where a(x) the tube area depends on the entry point x.

Solutions to Eq (8) could be plotted for each tile system, drainage rate and water content but a more compact solution is available if we change to dimensionless variables.

$X = x/S$ dimensionless distance between drains

$Z = z/S$ dimensionless depth

$A(X) = a/S^2$ dimensionless area

$T = Rt/\theta_S S$ dimensionless time

$\eta = S/D$ drain spacing to depth ratio (9)

With these substitutions Eq (8) becomes

$$T = \frac{A(X)}{\Delta X} \tag{10}$$

The advantage to Eq (10) is that T(X) is the same for all systems with the same ratio of drain spacing to barrier depth $\eta = S/D$. Figure 4 is a plot of dimensionless travel time T as a function of point of entry X from X = 0 (over the drain) to X = 1 (midway between drains). This result gives the travel time through the saturated zone. The total travel time should include the time to reach the water table (Eq (7)).

To illustrate the calculation using Eq (7) and Figure 4, we consider two examples.

EXAMPLE 1

Parallel drains buried 200 cm below the surface (d = 200 cm), spaced 5000 cm apart (S = 2500 cm) with a barrier 450 cm below the surface (D = 250 cm). Water content in unsaturated zone = 0.25 (cm^3/

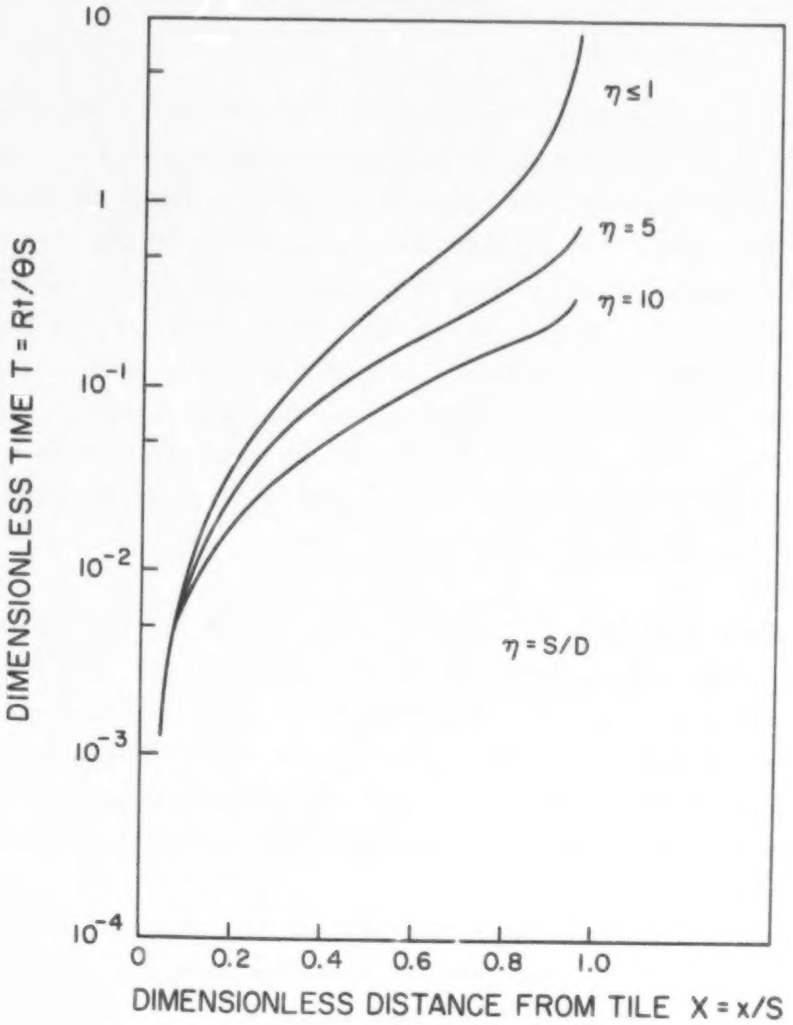

Figure 4. Dimensionless travel time $T = Rt/\theta S$ vs lateral point of entry X/S.

cm^3). Saturated water content $\theta_S = 0.35$ (cm^3/cm^3). Salt is leached from the surface horizon by sprinkler irrigation at $R = 5\ cm^3/cm^2/$ day. Calculate the time required for salt to reach the drain.

The travel time through the unsaturated zone (Eq (7)) is 10 days. In the saturated zone we consider entry points 250 cm, $X = .1$ and 2250 cm, $X = .9$ from the drain. Using Figure 4, $\eta = 10$, we find $T(.1) = 0.006$, $T(.9) = 0.15$. To find the time t we use Eq (9) $t = (\theta_S S/R)\ T = (175\ \text{days})T$ so $t(250\ \text{cm}) = 1.1$ day; $t(2250) = 26.3$ days.

Thus, salt leached from the surface near the tile ($X = 250$ cm) will reach the drain in 11.1 days, whereas salt originating far from the drain will require 36.3 days.

EXAMPLE 2

Same tile geometry ($d = 200$ cm, $S = 2500$ cm, $D = 250$ cm) with deep rooted crop grown in the unsaturated zone $\theta = .35$ of a fine textured soil $\theta_S = .50$. Consumptive use (ET) is 100 cm/year, and yearly irrigation is 150 cm (I). Rainfall is negligible. Calculate travel time for nitrate added with the irrigation water. The yearly drainage is equal to the irrigation minus consumptive use or $R =$ 50 cm/year.

Using the approximate relation (Eq (5)) for root zone travel time gives $t = .70$ year. In the saturated zone $t = (\theta_S S/R)\ T = (25\ \text{yr})\ T$. We again use the $\eta = 10$ curve to get $X(.1) = .006$, $X(.9) = .15$ so that $t(25\ \text{cm}) = .15$ year; $t(2250\ \text{cm}) = 3.75$ year.

Thus, nitrates leached from $x = 250$ cm will reach the tile in 0.85 year, whereas the arrival time from $x = 2250$ cm is 4.45 year.

These two examples show the versatility of dimensional analysis. One figure may be used for virtually all calculations, with soil, management, and drain geometry factors telescoped into the relation $t = (\theta_S S/R)\ T$ (Eq (9)) between real and dimensionless time.

Equation (9) suggests that the product $(\theta_S S/R)$ with dimensions of time, may be used along with Figure 4 to characterise the time response of an arbitrary system, with small values corresponding to small travel times (fast response) and vice versa. Table 1, taken from Jury (1975) gives response time parameters ($\theta_S S/R$) for a number of tile drain systems published in the literature.

References 5 and 7 are California tile lines with $\theta_S S/R$ from 20 to 1000 years, so that travel times will be extremely long for such systems. The last column gives the percent of salt initially in the soil that would be leached in one year, ranging from 4 to 100%.

Tile Drain Concentrations

Because of the variable distance and travel time to the drain, effluent concentrations represent a complicated average over the field. Some simple calculations will be used to illustrate this point.

EXAMPLE 1

Sudden change in salinity of the irrigation water from $C = 0$ to $C = Co$.

The effluent concentration will remain at $C = 0$ until the salt applied directly over the drain arrives at the tile. Calling this $T = 0$, we find that for $T > 0$ the effluent concentrations is a mixture of solution Co from those partsof the field with travel times less than T, and of pure water from the rest of the field. Thus

$$\mathrm{Cout}\,(T) = \mathrm{Co}X(T) \tag{11}$$

A plot of Eq (11) is given in Figure 5. For a California field with ($\theta_S S/R$) = 100 years and $\eta = 10$, the concentration will rise to 0.6 Co after 10 years.

EXAMPLE 2

Single pulse of concentration Co applied at the surface between $T = T_A$ and $T = T_B$. ($T_B - T_A = \Delta T$).

If we again call $T = 0$ the time when the pulse first reaches the tile then the output concentration for times $T_1 > 0$ is made up of solution concentration Co from that part of the field discharging

Table 1. Published tile-drainage studies and estimate travel time parameters ($\theta_S S/R$).

Source	Soil Type	Tile Spacing	Barrier Depth	Depth to Tile	Tile Discharge	$\frac{\theta_S S}{R}$	% Leached
		m			m/yr	yr	1st yr
1. Bolton et al (1970)	Clay	76.0	?	0.7	0.16	122	12
2. Calvert and Phung(1971)	Fine Sand	18.3	?	?	1.01	3.6	74
3. Erickson and Ellis(1971)	Clay Loam	20.0	?	1.0	0.03	157	8
4. Glandon (1971)	Several	60.0-80.0	?	1.5-3.0	<0.47>*	≥32	≤28
5. Letey et al (1977)	Several	>35.0	?	2.0	varied	20-100	14-40
6. Meek et al (1969)	Silty Clay Loam	60.0	?	1.8	0.48	31	30
7. Johnson et al(1965)	Loam	60.0-200.0	?	2.0	<.18>	32-1000	4-30
8. Sadler et al(1965)	Silt Loam	6.1	0.2	0.9	43.8	.025	100
9. Talsma (1967)	Sand	20.1	2.0	1.5	1.8	2.3	85
10. Talsma (1967)	Loam	26.8	2.0	1.5	5.3	1.0	98

*<>denotes average for all fields studied.

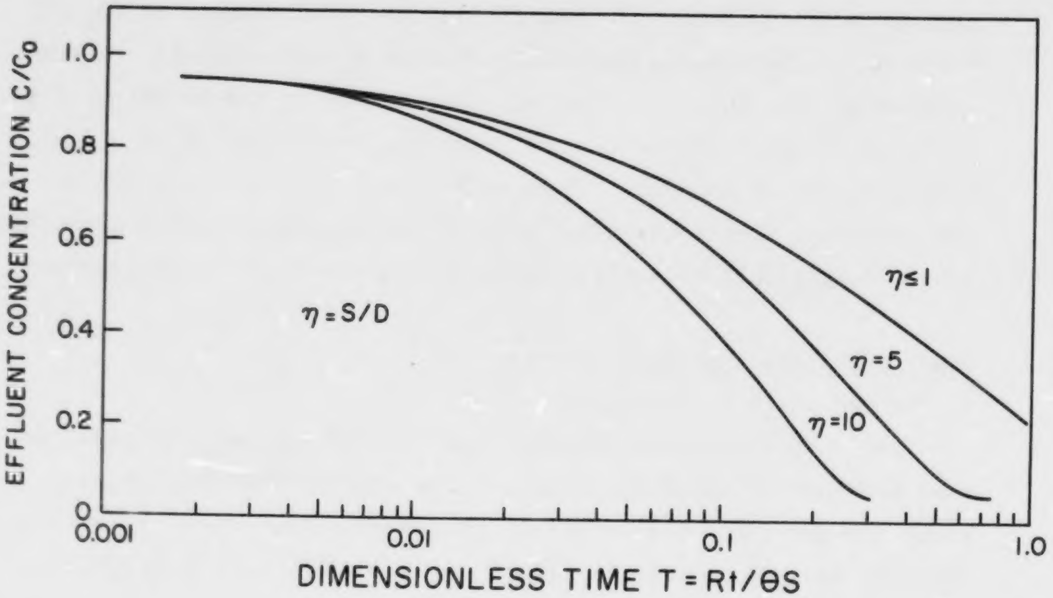

Figure 5. Output concentration vs dimensionless time T = Rt/θS for system leached with pure water.

the pulse and C = 0 from the rest of the field.

$$C(T_1) = Co\,(X(T_1) - X(T_1 - \Delta T)) \tag{12}$$

A plot of Eq (12) is shown in Figure 6 for the case $\Delta T = .01$. The pulse appears very spread out at the outlet, requiring a long time to move completely through the system. For example if $(\theta_S S/R) = 100$ years (California field) then Figure 6 represents $\Delta t = 1$ year of sending in solution of concentration C = Co at the surface. This might represent a single year of intensive fertilization of a field naturally low in nitrogen. Nine years later (T = .1) only 60% of the pulse has flowed through the tile, and residual nitrate from the single fertilization is still appearing in the drain concentration.

Other Artificial Drainage Systems

1. Ponded surface conditions

The main difference between this case and the one discussed is that the rate of water entry is not the same at different points along the surface. The result is that even longer travel times are required to enter the drain from large distances away from the tile (Jury, 1975).

2. Layered soil profile

For this system the streamlines will have different shapes than Figure 2, but they may still be calculated (Kirkham et al, 1974) and the rest of the procedure applied as above.

LIMITATIONS OF THE MODEL

Free Drainage

There are two major problems with the application of the piston flow model Eq (4). First, water does not move through the entire pore volume θ, particularly at high water contents. For this reason,

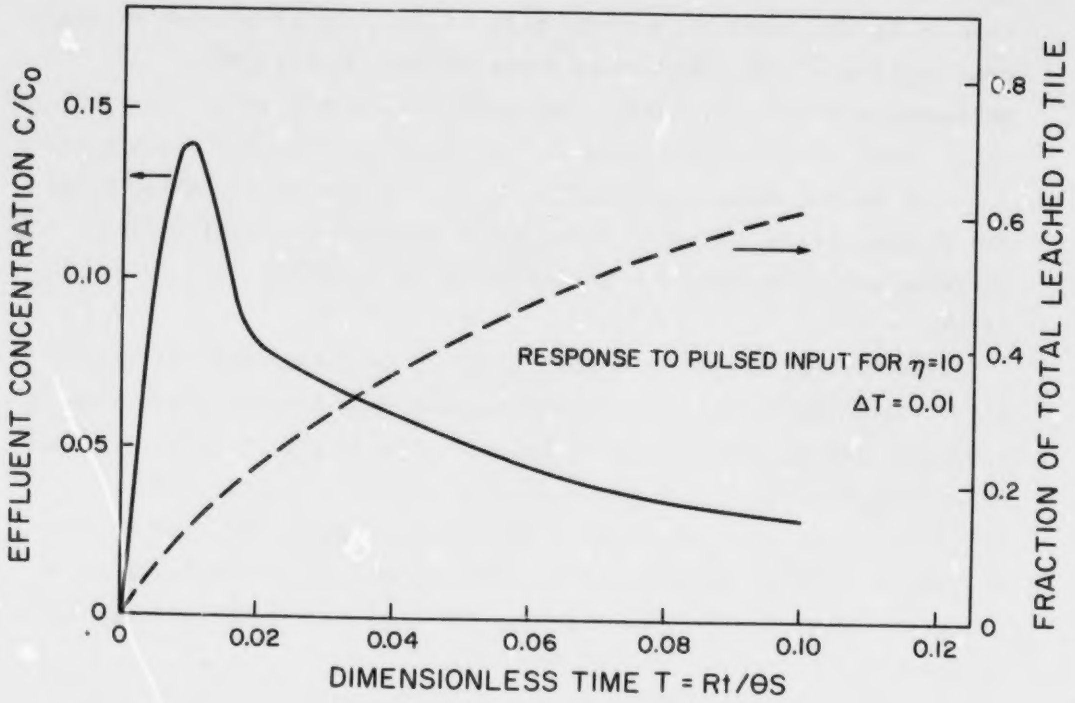

Figure 6. Output concentration C/Co vs dimensionless time T = Rt/θS for step input of solution C = Co during a time interval ΔT = 0.01.

the model often underestimates the movement of solution introduced at the soil surface, as observed by Warrick et al (1971), and Endelman et al (1974). For the same reason, solutes initially in the soil are not leached as rapidly by water introduced at the surface as Eq (5) predicts, because part of the salt is present in stagnant regions of the wetted pore space (Miller et al, 1965; Balasubramanian et al, 1973). Furthermore, certain soils, particularly clays, have preferential channels for water flow, so that travel times are much shorter than predicted by Eq (4) (Kissel et al, 1975; Tyler and Thomas, 1977). Finally, some soils show uneven infiltration patterns due to surface irregularities or instabilities (Jury et al, 1976).

The second problem in using Eq (4) is in estimating drainage flux. Although Pratt et al (1972) assumed that the ratio of chloride concentration at the surface to chloride concentration in the drainage equalled the ratio of the drainage flux to the irrigation flux, chloride values at a given depth are quite variable and lead to error in this prediction (Biggar and Nielsen, 1973). An alternative is to measure evapotranspiration directly, but this requires considerable equipment.

Artificial Drainage

Problems in the piston flow model for movement to drains are the failure of water to move through the entire pore space, nonuniform infiltration through cracks, channels, or backfill material, and neglecting diffusion and dispersion of salt within the soil solution. No studies have been made of the magnitude of these effects.

SUMMARY AND CONCLUSIONS

Use of the piston flow approximation for water and salt movement through soil leads to simple equations to estimate the soil residence time for nitrate between surface and groundwater or tile drain. These equations require knowledge of drainage or discharge rate, soil water content, location of tile drains, water table, and depth to layer of reduced permeability. For some systems, these equations will be inaccurate because of nonuniformities in water flow or substantial stagnant space in the wetted soil volume, but in many cases they will give a reasonable estimate of the phase lag between the application of a chemical at the soil surface and its subsequent appearance in a drain outlet or in underlying groundwater.

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USE OF MATHEMATICAL RELATIONSHIPS TO DESCRIBE THE BEHAVIOR OF NITROGEN IN THE CROP ROOT ZONE

J. M. Davidson and P. S. C. Rao[1]

ABSTRACT

A procedure to estimate the movement of water-soluble nitrogen species (NO_3 and NH_4) was developed by assuming that (i) the soil-water residing in all pore-sequences participates in the transport process, and that (ii) the soil-water initially present in the soil profile was completely displaced ahead of the water entering at the soil surface. Field-capacity and initial soil-water content distribution in addition to total water inputs were necessary parameters to estimate solute transport in the root zone. First-order kinetics were assumed to describe the nitrogen transformations (mineralization, immobilization, nitrification, and denitrification). These transformation processes were considered to occur under ideal conditions. Plant uptake of water and nitrogen (nitrate and ammonium) was estimated, respectively, from potential evapotranspiration and nitrogen uptake rate under ideal environmental conditions for a given crop. Actual plant uptake of water and nitrogen was dependent upon the available soil water and total mineral nitrogen within the crop root zone. These mathematical relationships could be solved using a programmable desk-top calculator; however, a larger computer was needed when more complex sub-models were employed to describe soil-water uptake. The proposed mathematical relationships can provide field managers and regulatory personnel with an integrated description of the behavior of nitrogen in the root zone during a crop growing season.

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INTRODUCTION

Nitrogen is an essential element in all biological systems. In an undisturbed soil environment, the nitrogen cycle is closed and the total nitrogen content of the system (soil plus plants) does not change with time. This delicate balance is disturbed, however, when man uses the soil for food production and separates the site of nitrogen consumption (metropolitan areas) from where it is assimilated (plants and animals). Because of this, it has been necessary to supplement most agricultural soils in recent years with nitrogen fertilizer. The inorganic nitrogen added in this manner represents only a small part of the total amount of nitrogen contained in a soil since most soil nitrogen is organically bound. However, it is the inorganic nitrogen which is the most dynamic and of interest when evaluating environmental contamination and agricultural production potential.

The fate of various nitrogen forms at and below the soil surface is governed by a variety of interrelated and complex processes. These processes involve inorganic (NH_4, NO_3, NO_2, N_2O, and N_2) and organic nitrogen forms which exist simultaneously in the soil. These and other nitrogen substrates undergo continuous reversible and/or irreversible transformations owing to chemical and microbiological processes. The water-soluble nitrogen species (NH_4, NO_3, and NO_2) are also translocated through the soil by molecular diffusion and more importantly by soil water. Ammonium (NH_4) and nitrate (NO_3) concentration in the soil is complexed further by absorption by plant roots. The extent of water and nitrogen uptake by plants is determined, in part, by the transpiration demand, which in turn is dependent upon plant species, growth stage, and meteorological conditions.

The complexity of the soil-water-plant system is enhanced further by the fact that all of the above processes are transient in nature and occur simultaneously. Therefore, a prerequisite to describing the fate of inorganic nitrogen forms in soil-water-plant systems is a complete understanding of the nitrogen transformation and transport processes. A considerable amount of information is available regarding soil nitrogen and its behavior under laboratory and field conditions (Bartholomew and Clark, 1965). However, due to the nature and conditions under which much of the research was conducted, it is difficult to combine the results from these individual experiments into a form that can be used to develop relationships for simulation and/or prediction purposes.

Mathematical relationships to describe physical, chemical, and biological processes are generally of three types. Stochastic relationships assume the processes being described obey or follow the laws of probability. Empirical relationships are developed from experience and observations and regression equations are used to correlate known inputs (temperature, rainfall, nitrogen, etc.) with measured outputs (yield, nitrogen leached, etc.). Mechanistic relationships are based upon established physical, chemical and biological laws that describe specific processes. Mechanistic relationships are versatile in that extensive historical records are not necessary for their application at a given site; however, these relationships require a complete understanding of processes. The general limitation of mechanistic relationships, therefore, is an inadequate understanding of the system. Empirical relationships are of assistance in such situations because major variables have been identified as well as their influence on the system.

The mathematical relationships used in this manuscript to describe the movement, transformations, and plant uptake of various nitrogen forms in an unsaturated soil profile are mechanistic in concept, but empirical in development and presentation. A minimum amount of soil input information is needed for these relationships to obtain an integrated description of the various nitrogen forms in a plant root zone. The benefits of having a procedure for estimating the behavior of various soil inorganic and organic nitrogen forms with time are thought to outweigh the loss in accuracy resulting from this simplified approach.

SOIL-WATER AND NITROGEN MOVEMENT

As the soil solution moves through the interconnecting pore sequences of a soil, nitrate and ammonium ions as well as other soluble substances are carried with it. To describe the actual distribution of these solutes in a soil profile requires complex mathematical equations and analytical procedures. However, computational procedures to estimate only the position of a nitrate or ammonium front entering the soil at the soil surface and moving vertically down through the profile in response to different water inputs are easier and may be of equal benefit. This type of computation requires the assumptions that (i) the soil solution residing in all pore sequences participates in the transport process, and (ii) the soil solution in the profile is displaced ahead ("pushed ahead") of the applied water. These assumptions were also used by Frere (1975) to predict water and sediment transport from watersheds. Exceptions to the assumption that all pore-sequences participate in the transport process are discussed by Rao et al (1974) and Quisenberry and Phillips (1976); however, for many cases the assumption appears reasonable as demonstrated by Rao et al (1976b).

Water ponded on a soil surface fills all the pores as it moves down through the soil profile. The location of the wetting front (sharp boundary between the "wet" and "dry" soil) below the soil surface is related directly to the amount of water that has entered the soil. If a fertilizer containing nitrate is broadcast over the soil surface prior to applying the water, nitrate will enter the soil during infiltration and move with the water. Thus, the applied water and nitrate move together because nitrate is an anion and is not significantly influenced by the positively-charged clay particles in most soils.

To estimate the location of the wetting front for various amounts of infiltrated water, consider the following mathematical relationship:

$$d_{wf} = \frac{I}{\theta_w - \theta_i} \qquad (1)$$

where I is the amount of infiltrated water (centimeters or inches), θ_w the average volumetric soil-water fraction (cm^3/cm^3) in the wetted zone behind the wetting front, and θ_i the volumetric soil-water fraction (cm^3/cm^3) in the soil prior to adding the water and d_{wf} the distance from the soil surface to the wetting front.

Using equation 1 to calculate the position of the wetting front after 5 and 10-cm of water have entered a dry soil ($\theta_i = 0$) places the front at 14.3 and 28.6-cm if the soil-water content behind the wetting front is taken as 0.35 cm^3/cm^3 (Figure 1). However, if the initial soil-water content (θ_i) were 0.1 cm^3/cm^3, the wetting front would be located at 20 and 40-cm below the soil surface (Figure 2). The difference in wetting front position for the same input arises from the fact that the initial water in the previously wet soil was displaced ahead (assumption (ii) in the previous section) of the infiltrating water. Thus, the soil water between the 14.3 and 20-cm soil depth (Figure 2) is water that was in the soil prior to the 5-cm water application. From this it is concluded that the 5-cm of water added to the wet soil moved to approximately the same depth as that added to the dry soil ($\theta_i = 0$). This phenomenon can be observed where discharge from drain tiles frequently begins shortly after the initiation of water application to the soil surface. The water in the tiles is that displaced from the soil by the infiltrating water. This concept is important when determining the depth to which nitrate and ammonium will move in the soil.

The initial soil-water content of a field soil is generally not uniform, but dry near the surface and wet deeper in the profile. Although more complex, this case does not alter the previous discussion or conclusions. Using a programmable desk-top calculator, the amount

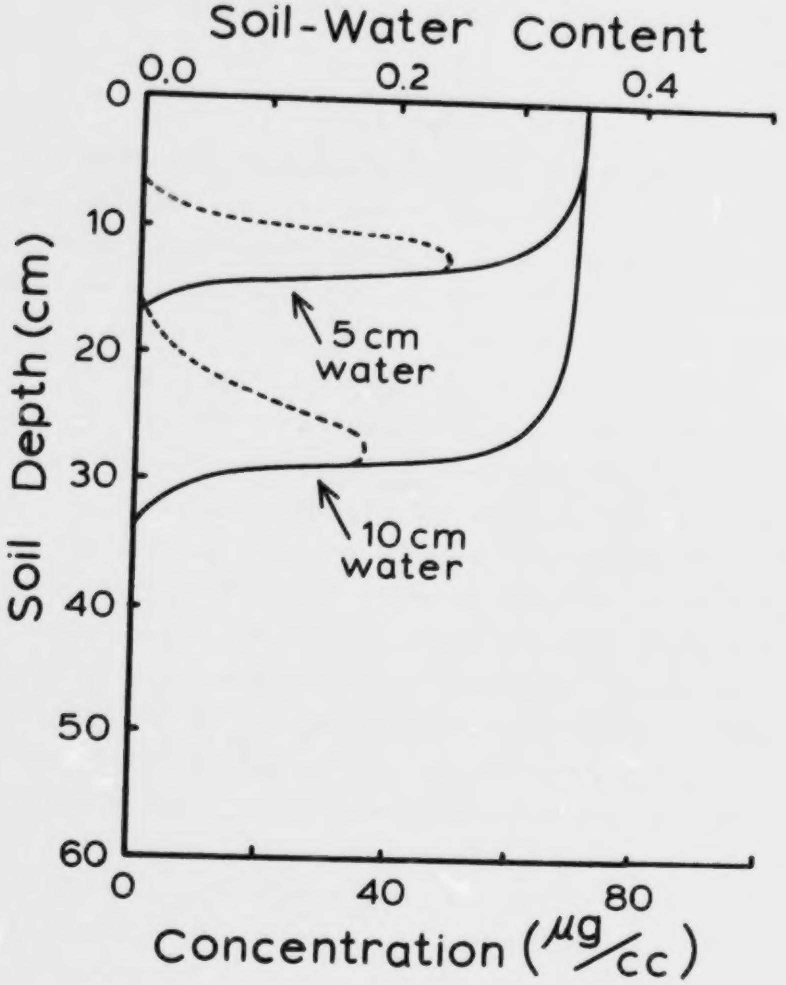

Figure 1. Distribution of soil-water (solid lines) and nitrate (dashed lines) after 5 and 10-cm of water had infiltrated into an initially dry (θ_i=0.0) soil profile (from Rao et al, 1976b).

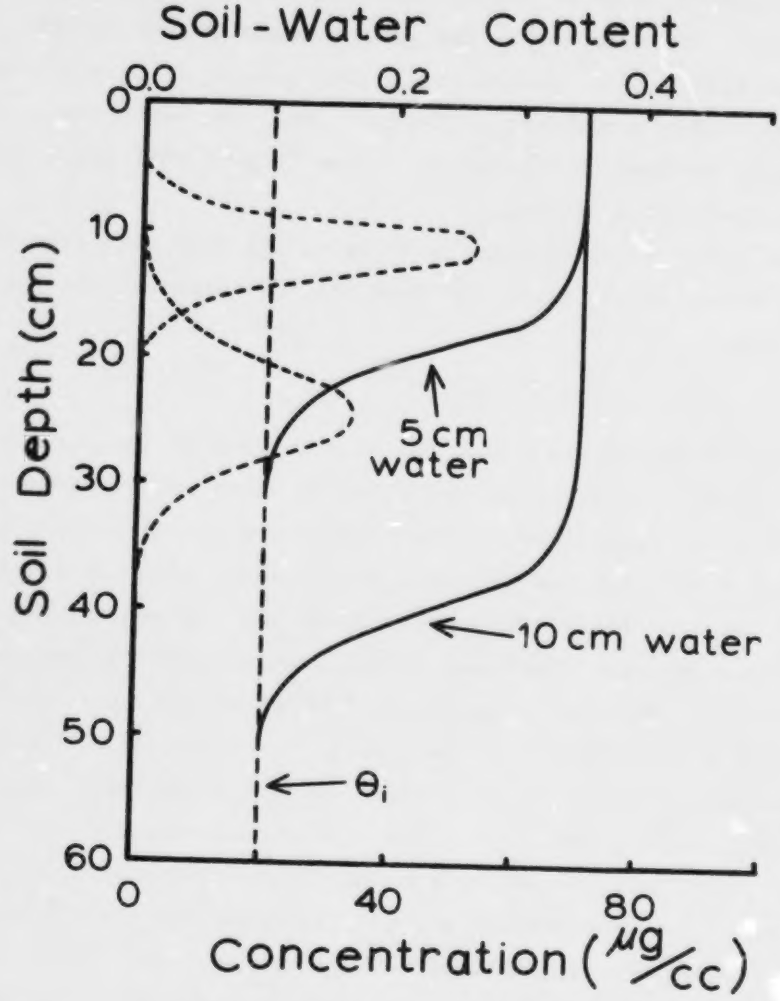

Figure 2. Distribution of soil-water (solid lines) and nitrate (dashed lines) after 5 and 10-cm of water had infiltrated into a soil profile at a uniform initial soil-water content (θ_i shown as vertical dashed line) of 0.1 cm^3/cm^3 (from Rao, 1976b).

of water pushed ahead of the infiltrating water for this case can be calculated. For example, if the water content behind the wetting front was 0.35 cm^3/cm^3 following the application of 5-cm of water to the initially nonuniform soil-water case, the infiltrating water would still be located between the 0 and 14.3-cm soil depth with the wetting front below 14.3-cm.

The position of the nitrate entering the soil with the infiltrating water can be calculated from the following mathematical relationship:

$$d_{nf} = \frac{I}{\theta_w} \qquad (2)$$

where d_{nf} is the distance from the soil surface to the nitrate front and the other symbols are as described earlier. Note that the position of the nitrate front depends only upon the amount of water that has infiltrated the soil and the average volumetric soil-water content behind the wetting front. This means that the soil water containing the nitrate solution displaces the soil water initially present in the soil profile. This is illustrated in Figures 1 and 2 where the nitrate pulse from the fertilizer is indicated by a dashed line. Note that in both cases (Figures 1 and 2), the nitrate pulse front is located at 14.3 and 28.6-cm after 5 and 10-cm of water had entered the soil. The nitrate front lags behind the wetting front for the previously wet soil case; thus, the wetting and nitrate front are not analogous.

Following the cessation of infiltration the soil-water content behind the wetting front decreases owing to drainage or redistribution. This process results in a deeper penetration of the water and nitrate front into the soil profile. Following two to three days of drainage, the change in soil-water content with time behind the wetting front become small and a soil-water content commonly referred to as "field capacity" (θ_{fc}) is said to have been reached. The nitrate movement that occurs because of redistribution can be estimated from the amount of "drainable" water (θ_w - θ_{fc}) between the soil surface and the nitrate front (d_{nf}). Fine textured soils generally have a field capacity soil-water content two to three times larger than that of coarse textured soils; thus, they have a smaller amount of drainable water. The distance the nitrate front will move owing to redistribution is considerably less for fine compared to coarse textured soils.

The position of the nitrate front following infiltration and redistribution of the soil-water to field capacity can be calculated from the following mathematical relationship:

$$d_{nf} = \frac{I}{\theta_{fc}} \qquad (3)$$

The use of equation (3) is limited to cases where the nitrate front originates at the soil surface. If the nitrate front is located at some depth in the soil prior to the application of water, then the amount of drainable water above the nitrate front after infiltration must be determined. This can be done by knowing the initial soil-water content distribution above d_{nf}, θ_w, and θ_{fc}. These calculations can be made on most programmable desk-top calculators.

The retardation of the ammonium front due to ion-exchange with the soil can be calculated from a knowledge of the adsorption characteristics for ammonia with a given soil. The retardation would be greater for fine than for coarse textured soils (Rao and Davidson, 1978). The retardation coefficient (R) is proportional to the soil bulk density (p), magnitude of adsorption (K), and inversely proportional to volumetric soil-water content (θ). This relationship may be expressed as:

$$R = (1 + \frac{pK}{\theta}) \qquad (4)$$

Note that, for a given K and p, the retardation coefficient increases with decreasing θ. Thus, retardation of ammonium movement is greater in a drier soil than in a wet soil. The ammonium front location is calculated by dividing the nitrate front location (see Eq. 3) by the retardation coefficient (R).

NITROGEN TRANSFORMATIONS

As stated earlier, nitrogen exists simultaneously in various inorganic and organic forms in the soil. These nitrogen forms undergo continual transformations from one form to the other. Both microbial and chemical processes are responsible for these transformations. Some of the important nitrogen transformation processes in soils are: mineralization, immobilization, nitrification, and volatilization. In cases where urea fertilizers are added to the soil, hydrolysis of urea to ammonium must also be considered. The interrelationships between these nitrogen transformation processes in soils are schematically illustrated in Figure 3.

Mineralization process is the microbiological conversion of organic nitrogen to ammonium. This process is also referred to as ammonification. Immobilization is the opposite to mineralization, since this process involves microbial assimilation of inorganic nitrogen into organic forms. Thus, when both these processes are occurring simultaneously in a soil, only the net result may be observed. A net increase in the inorganic nitrogen content is evident when mineralization prevails, while a net decrease in the inorganic nitrogen content results when immobilization is predominant. Although large supplies of organic nitrogen exist in soils, only a small fraction of it is available to the micro-organisms as an energy source. This fraction is referred to as the potentially mineralizable organic-nitrogen and constitutes about 10% of the total soil organic-nitrogen (Stanford and Smith, 1972). On the other hand, potentially mineralizable fraction of the organic-nitrogen in many common organic wastes (e.g., animal manure, liquid dairy waste, etc.) has been estimated to range from 50 to 90% (Reddy et al, 1977). It should be recognized that mineralization is the predominant soil process responsible for the net production of ammonium from organic forms.

Nitrification is the microbiological oxidation of ammonium to

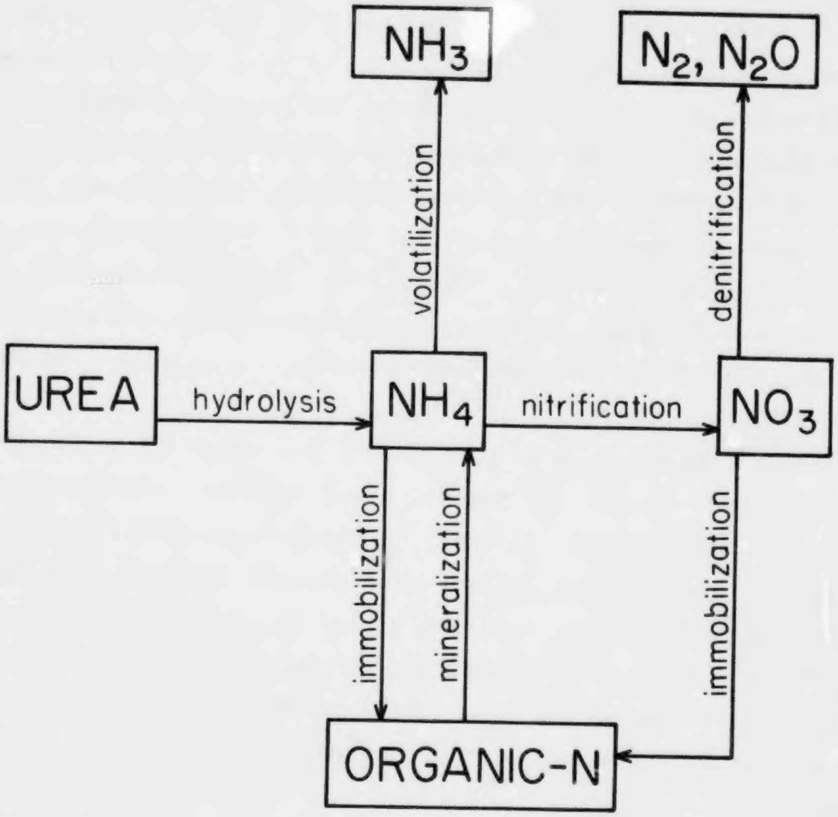
NH$_3$
N$_2$, N$_2$O
volatilization
denitrification
UREA
hydrolysis
NH$_4$
nitrification
NO$_3$
immobilization
mineralization
immobilization
ORGANIC-N

Figure 3. Schematic representation of the inter-relationships among the major nitrogen transformation processes in soils.

nitrate, with the formation of nitrite as an unstable intermediate product; under field conditions, nitrite is rapidly oxidized to nitrate. Hence, nitrification may be considered as a single-step process with the ammonium to nitrite conversion step controlling the rate of nitrification. Nitrogen is lost from soils under certain conditions in gaseous forms due to volatilization of ammonium and denitrification of nitrate. It is known that ammonia volatilization losses increase with increasing soil pH (alkaline conditions). Since microorganisms responsible for denitrification are anaerobic in nature, gaseous losses of nitrogen by denitrification increase with increasing soil-water content. Volatilization and denitrification processes are discussed in detail by Rolston (1978), whereas nitrification, mineralization, and immobilization are discussed by Broadbent (1978).

Most of the nitrogen transformation processes are mediated by soil microorganisms. Hence, the rate at which a given transformation process proceeds is determined by the size of the population and/or the activity of specific soil microorganisms. Several soil environmental factors (pH, soil-water content, temperature, and aeration) are known to significantly influence nitrogen transformations. Soil temperatures in the range of 30-35C and a neutral soil pH appear to be optimal for most microbial populations controlling nitrogen transformations in soils.

Several researchers have reported that nitrogen transformations were first-order processes, i.e., the reaction rate was directly proportional to the concentration of the nitrogen species undergoing the transformation. These proportionality constants are referred to as the rate coefficients and are symbolized as k. The values of k depend upon several soil factors. It should be recognized that large values of k suggest rapid transformations. The time required for 50% of the initial amount of a given nitrogen species to be transformed is defined as the half-life ($t_{\frac{1}{2}}$). Assuming a first-order process, the values of $t_{\frac{1}{2}}$ can be calculated given a k value. Estimates of $t_{\frac{1}{2}}$ for various nitrogen transformations in soils, based on published data, are listed in Table 1. On this basis, hydrolysis of urea appears to be the fastest reaction, while mineralization-immobilization processes are the slowest.

Table 1. Values of half-lives ($t_{\frac{1}{2}}$) for selected nitrogen transformation processes in soils, estimated on the basis of published data.

Transformation Process	Half-life, $t_{\frac{1}{2}}$ (days)
Mineralization * (org N $\rightarrow NH_4$)	100-200
Nitrification ($NH_4 \rightarrow NO_3$)	10-20
Volatilization ($NH_4 \rightarrow NH_3$)	20-50
Denitrification ($NO_3 \rightarrow N_2 + N_2O$)	7-15
Urea Hydrolysis (Urea $\rightarrow NH_4$)	3-7

* Net Mineralization

Mathematical relationships for describing various nitrogen transformations in soils have been proposed by several investigators (Mehran and Tanji, 1974; Endelman et al, 1973; Rao et al, 1976a; Selim et al, 1978). These relationships are complex and sophisticated computers are required to solve them. However, simpler equations to calculate the net rates of production or loss may be developed when only the total amounts of each nitrogen species in the profile is considered. These rate equations and their analytical solutions, adaptable for use with programmable desk-top calculators, were described in detail by Davidson et al (1978) and will be used in this manuscript. Principal limitations of this approach are: (i) only total amounts of each nitrogen species were considered and not their concentration distributions in the soil profile, and (ii) the transformation rate coefficients were assumed to not be influenced by soil environmental factors. Furthermore, these relationships are applicable only when denitrification, volatilization, and nitrate immobilization (see Figure 3) are insignificant. Since the mathematical relationships considered in this manuscript are designed to provide an integrated description of nitrogen transformations, the above stated limitations are not expected to be too severe, especially for well-drained soil.

PLANT UPTAKE OF WATER AND NITROGEN

The inherent complexity of the crop root zone and the dynamic nature of the plant's uptake of water and nutrients defies an exact mathematical description. Furthermore, an understanding of the physiological characteristics of crops and their needs for water and nutrients is at best incomplete. This necessitates a gross simplication of the actual processes involved in plant uptake when developing empirical relationships for predictive purposes.

In a macroscopic approach, the entire root system may be considered as a sink for nitrogen and water. The crop demand during the growing season then represents the sink strength, while the distribution of this sink in the soil profile is represented by the root distribution. In this approach, the absorption capacity of the roots for water and nitrogen is assumed uniform over the entire root system. Thus, the rate of crop uptake for water (Q_W) and nitrogen (Q_N) may be expressed, respectively, as cm water/cm root/day and μg nitrogen/cm root/day.

The rate of water uptake by plant roots is governed by (i) the ability of the soil to conduct water to the roots, and (ii) the plant water demand rate as determined by the transpiration rate. The hydraulic conductivity ($K(\theta)$) of the soil is dependent on the soil-water content and in most soils exponentially decreases as the soil-water content decreases. Therefore, plants growing in a dry soil may undergo wilting if the water transport rate to the roots is less than the transpiration demand. Molz and Remson (1970) have proposed an empirical relationship to describe the dynamics of water extraction by plant roots. This approach takes into account the factors discussed above. Using their relationship, the transpiration demand for water is distributed over the entire root system and is proportional to the product of $K(\theta)$ and $R(z,t)$, where $R(z,t)$ is the root length distribution.

Nitrogen uptake by plant roots involves the movement of water-soluble nitrogen species (ammonium and nitrate) to roots followed by their absorption across the root surfaces. Convective flow of soil-water towards roots in response to transpiration results in the mass-flow of ammonium and nitrate along with the soil water. The concentration of these ions at or near the roots decreases when the rate of uptake exceeds the rate of their supply by mass flow. Diffusion of nitrate and ammonium towards roots in response to this concentration can then take place.

Although arguments abound in the literature as to the relative importance of these two processes, it is generally agreed that when mass-flow is restricted (such as due to moisture stress), nitrogen uptake due to diffusion can be significant. Furthermore, the relative importance of mass-flow and diffusion depends upon the geometry of the root system. Higher root densities result in shorter distances over which the ions must be transported; hence, diffusive supply of nutrients to roots can be substantial (Barley, 1970; van Keulen et al, 1975). Russel and Shone (1972) have demonstrated that when part of the intact root system of barley was exposed to solutions of higher nitrogen concentrations than the remainder, root proliferation was limited exclusively to that zone in which more favorable conditions existed. Thus, higher root densities would result in a greater nutrient uptake from the favorable zone in order to compensate for the remainder of the root system. Jungk and Barber (1974, 1975) have also arrived at similar conclusions based on their experiments. On the other hand, Brower and deWit (1969) have observed that root densities increased when nutrient supply was limited.

It is apparent from the foregoing discussion that our present understanding of the physiology of the plant root system and its ability for compensatory growth and uptake under conditions of nutrient and/or water stress is limited. However, the principal factors that govern nitrogen uptake (and that of other nutrients) by plants appear to be: (i) concentration of the nutrient in soil-water, and (ii) the root density (number, length, area, etc.) at a given depth in the soil profile. Several researchers have proposed empirical relationships, where the rate of nitrogen uptake (Q_N) remained essentially constant over a broad concentration value and exponentially decreased at low concentrations. Such relationships are based on analogies to enzyme kinetics and are generally referred to as Michaelis-Menton relationships. A graphical representation of the dependence of Q_N on the total amount of nitrogen is shown in Figure 4. Based upon the work of Dibb and Welch (1975), it will be assumed here that both ammonium and nitrate are taken up by plants. Data are presently unavailable to determine the fractional uptake of ammonium and nitrate when both species are present in the root zone. Hence, it will be assumed that, the rate of uptake of each species is proportional to its fraction of total inorganic-nitrogen. Based upon these assumptions, the following relationships may be stated:

$$Q_N = Q_1 + Q_2 = Q_{max} \frac{T_N}{K_N + T_N} \qquad (5)$$

$$Q_1 = Q_N \left(\frac{T_1}{K_N + T_N}\right) \qquad (6)$$

$$Q_2 = Q_N \left(\frac{T_2}{K_N + T_N}\right) \qquad (7)$$

where, Q is rate of plant uptake, T is total amount of a given nitrogen

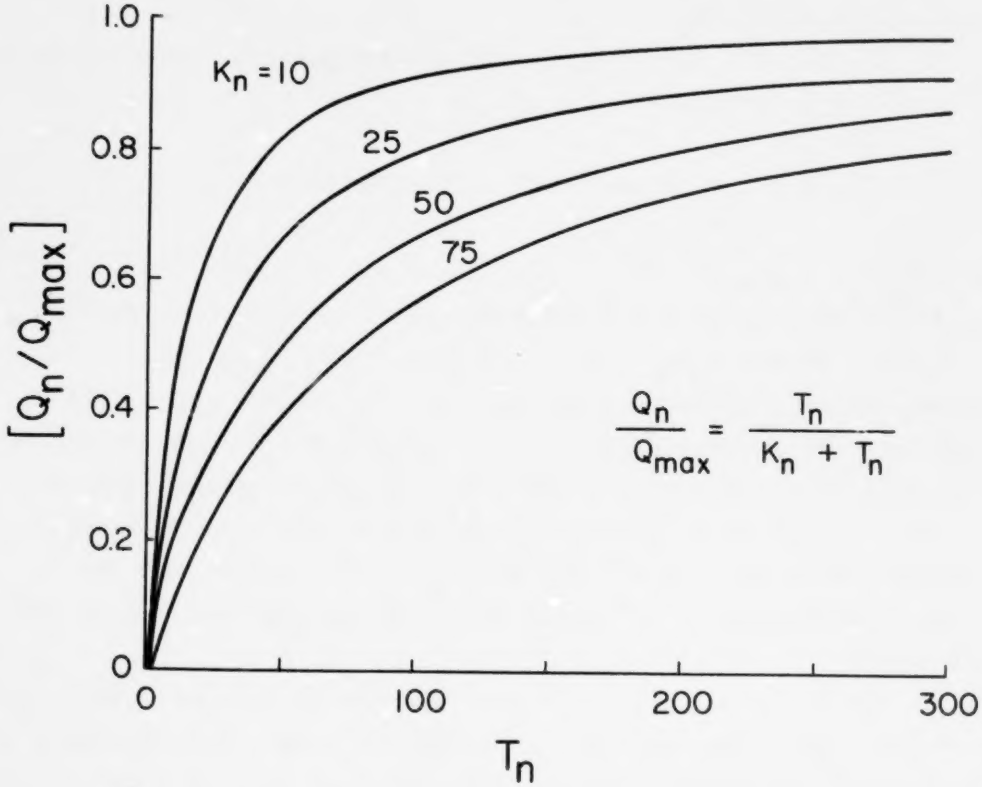

Figure 4. Graphical representation of the dependence of the rate of nitrogen uptake (Q_n) on the total amount of mineral nitrogen (T_n) in the crop root zone.

species, and subscripts 1, 2, and N refer to, respectively, ammonium, nitrate, and total inorganic-nitrogen (nitrate + ammonium). Also, K_N is an empirical constant that determines the rate at which Q_N diminishes with decreasing T_N (See Figure 4), and Q_{max} is the crop uptake demand for nitrogen under ideal growth conditions. Note that Eqs. (5)-(7) allow for simulation of reduced uptake under nitrogen "stress" conditions.

SIMULATION OF FIELD EXPERIMENTS

Case 1

NaNagara et al (1976) performed field experiments to measure nitrogen uptake by corn during a growing season in Kentucky. In addition to measuring nitrogen accumulation in corn plants, these authors also collected data on root length distributions in the soil profile, evapotranspiration water losses, rainfall/irrigation water inputs, and nitrogen fertilizer additions to the crop throughout the season. These experimental data were utilized to evaluate the empirical relationships referred to in the previous sections of this manuscript.

Calculated and measured cumulative nitrogen uptake by corn during the growing season are compared in Table 2. Reasonable agreement between calculated and measured uptake, especially for the total during the 34-97 day period, is encouraging; the differences are small considering the gross simplifications used in performing the calculations. The total amounts of ammonium, nitrate, and organic-nitrogen remaining in the corn root zone, as a fraction of that present initially, during the growing season are shown in Figure 5. These curves were calculated using the mathematical relationships for describing the nitrogen movement, transformations, and uptake. It is apparent that net mineralization of organic-nitrogen was small, while most of the ammonium was rapidly oxidized to nitrate. The total amount of nitrate in the crop root zone increased during the first 20 days (Figure 5) in spite of plant uptake, suggesting that the rate of nitrification exceeded that of nitrate uptake by the plant. However, beyond 30 days, the amount of nitrate decreased rapidly as nitrate became the major source for plant nitrogen due to depletion of ammonium. A total of 18.7-cm of water was received as rainfall during the corn growing season, whereas the accumulated transpiration losses were 23.9-cm. Thus, leaching losses of nitrogen were insignificant.

Table 2. Comparison of calculated and measured nitrogen uptake by corn grown under field conditions (Adapted from Davidson et al, 1977).

Growth Period (Days)	Nitrogen uptake (mg nitrogen/plant)		
	Measured*	Calculated	% Error
34-49	1435	1928	+34%
49-76	1101	1948	+77%
76-97	1496	683	-54%
TOTAL (34-97)	4002	4559	+14%

* Measured data was taken from NaNagara et al (1976).

Case 2

Watts and Hanks (1978) conducted field experiments to design efficient management practices for irrigation and nitrogen fertilization of irrigated corn on sandy soils of Nebraska. As a part of these studies, cumulative nitrogen uptake by corn and cumulative leaching losses of nitrogen beyond the root zone were measured throughout the season. Three levels of irrigation were considered: 0.8 evapotranspiration (ET) (low irrigation), 1.15 ET (medium irrigation) and 1.5 ET (high irrigation). All treatments received 250 kg nitrogen/ha as ammonium nitrate (broadcast) at planting. Other treat-

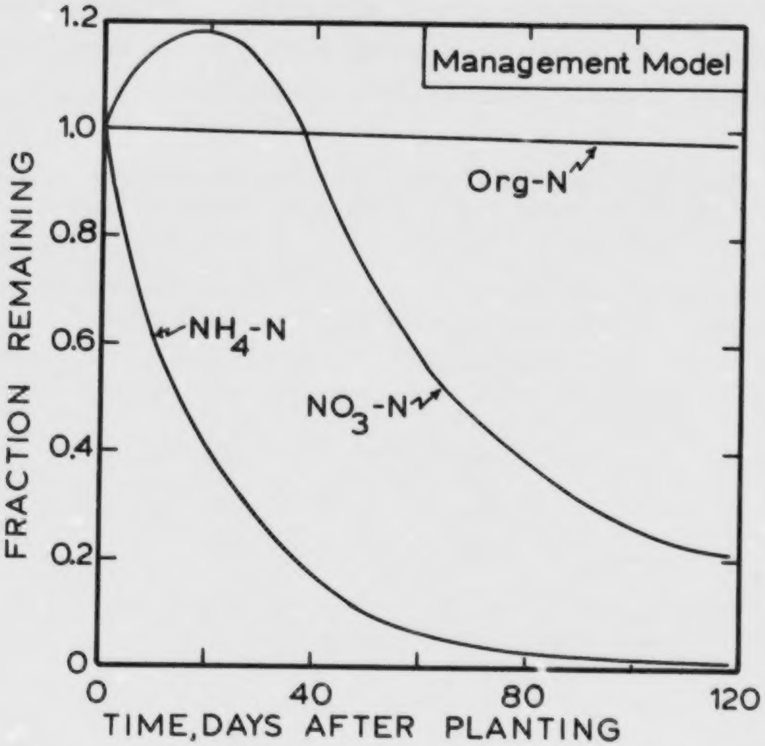

Figure 5. Fraction of applied nitrogen remaining in the plant root zone of Maury soil, simulation by the mathematical relationships, during the corn growing season (from Davidson et al, 1978).

ments were initiated by Watts and Hanks (1978), but only the above will be considered in this manuscript. Nitrogen balance sheet for the above cases for the end of the corn growing season, prepared from measured data and simulations is compared in Table 3. The calculated values were obtained using the previously described procedures and processes and input data provided by Watts and Hanks (1978). The unaccounted for nitrogen (measured data) is that necessary to obtain a mass balance from the field measurements. Factors such as nitrogen losses due to denitrification and volatilization in addition to experimental errors associated with sampling are probably responsible for the unaccounted-for nitrogen. Agreement between measured and calculated values is reasonable. The mathematical relationships used to obtain the calculated nitrogen data in Table 3 can also be used to calculate a nitrogen balance for other times during the growing season.

Although the procedure described and used in this manuscript to calculate nitrogen balances during the growing season requires several simplifying assumptions, it appears to give a reasonable first-order approximation. The procedure requires further testing and evaluation, but may be of some immediate benefit to field personnel not having another alternative. An example of such use is described in the next section. Most desk-top programmable calculators could be programmed to make the necessary calculations.

SIMULATION OF THE EFFICIENCY OF MANAGEMENT PRACTICES

The empirical relationships developed in this manuscript were also used to simulate three irrigation application schemes in order to examine their relative efficiencies in maximizing plant uptake and thereby reducing leaching losses beyond the crop root zone. The soil parameters were chosen to represent a well-drained homogeneous, and deep sandy soil profile, while the crop parameters were for corn.

Table 3. Comparison of nitrogen balance sheet prepared from measured data and that calculated using the mathematical relationships described in this manuscript. Measured data are for corn grown under three irrigation management schemes (Data of Watts and Hanks, 1978).

	Low Irrigation (0.8 ET)		Medium Irrigation (1.15 ET)		High Irrigation (1.5 ET)	
	Measured	Calculated	Measured	Calculated	Measured	Calculated
	(kg nitrogen/ha)		(kg nitrogen/ha)		(kg nitrogen/ha)	
Initial Mineral-nitrogen	58	58	55	55	47	47
Fertilizer-nitrogen added	250	250	250	250	250	250
Mineralized from Organic-nitrogen	84	86	84	77	84	76
Plant uptake	178	200	160	206	153	186
Mineral-nitrogen in Profile at Harvest	38	132	44	39	26	6
Leaching Loss	60	62	73	136	120	182
Unaccounted	116	--	112	--	83	--

Two water management schemes simulated were: (i) natural rainfall with no supplemental irrigation, and (ii) controlled amounts of irrigation under no rainfall conditions. The amount of irrigation applied was equal to or 1.5 times greater than the transpiration demand. Irrigation was allowed only when plant available water in the root zone was less than 60% of the total available water. The rainfall data used as input in scheme (i) was obtained from weather records (for May-Aug., 1974) maintained at the Agronomy Research Farm, University of Florida, Gainesville. A single broadcast application of ammonium nitrate fertilizer at 300 kg nitrogen/ha was applied at planting.

The position of the nitrate pulse in the soil profile during the growing season, simulated for three water application schemes, is presented in Figure 6. Also shown is the progression of maximum depth in the soil profile to which plant roots had grown. The nitrate pulse resides well within the crop root zone during the entire season for the case in which the amount of irrigation water was equal to that depleted by the crop (curve labeled 1.0 ET in Figure 6). For the case in which the amount of irrigation water applied was 1.5 times that required by the crop (curve labeled 1.5 ET in Figure 6), the nitrate pulse was leached beyond the root zone after about 65 days. The intensity and frequency of the rainfall events chosen here were such that the nitrate pulse was leached rapidly out of the root zone very early in the season (only five days after planting) as indicated by the curve labeled "rainfall" in Figure 6. Such observations are not uncommon in field studies involving sandy soils in Florida. A major rainfall event of approximately 5-cm can move the fertilizer nitrogen essentially out of the plant root zone.

The effect of simulated water application management schemes on the cumulative nitrogen uptake by corn is illustrated in Figure 7. The "ideal" uptake demand for nitrogen was met under the 1.0 ET treatments at all times. This was possible since the nitrate pulse

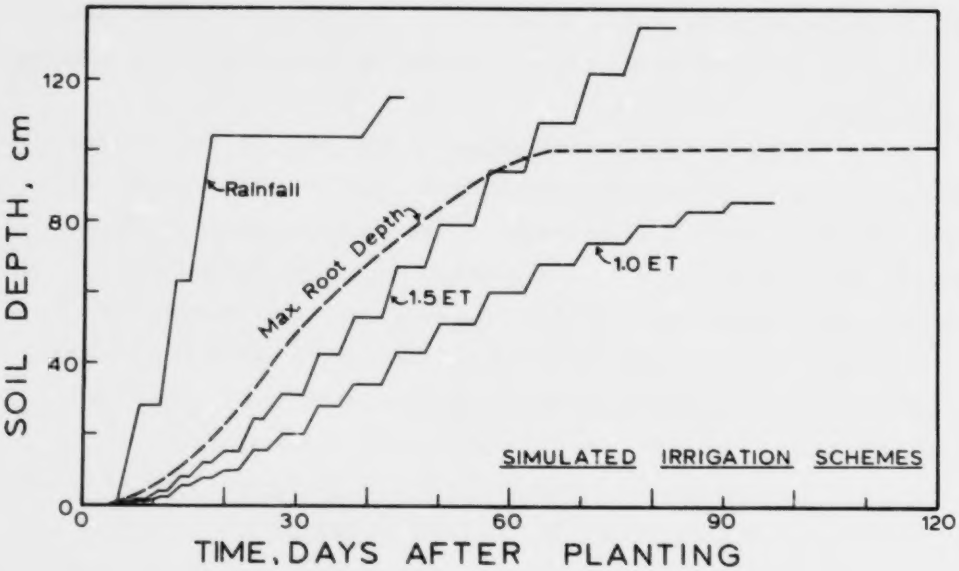

Figure 6. The predicted depth of nitrate front under three water application schemes during the growing season in a sandy soil planted to corn. Increase in the maximum depth of rooting (L) with time is shown as the dashed line (from Davidson et al, 1978).

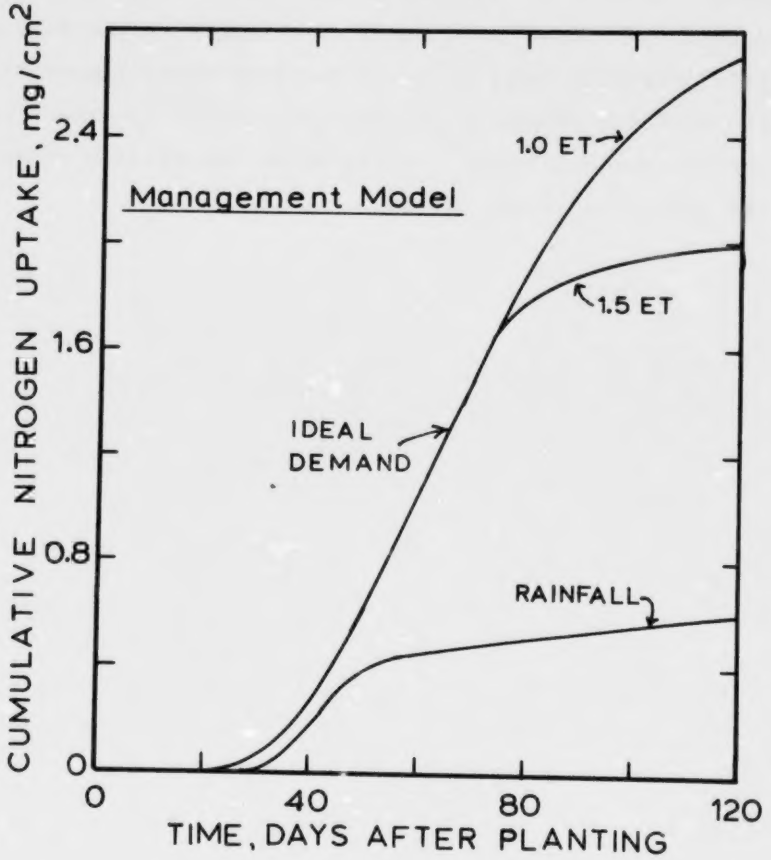

Figure 7. Cumulative nitrogen uptake by corn grown in a sandy soil under three water application schemes, as simulated by the mathematical relationships described in this manuscript (from Davidson et al, 1978).

resided within the root zone and was available to roots for absorption in sufficient quantities. The "ideal" demand, however, was not satisfied for the 1.5 ET treatment; the time at which this curve deviated from the "ideal" (Figure 7) corresponds to the time when the nitrate pulse was leached out of the root zone (Figure 6). The cumulative nitrogen uptake curve for the rainfall treatment deviates significantly from the "ideal" curve, since the nitrate pulse was beyond the root zone at all times.

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DIAGNOSTIC TECHNIQUES USED TO IDENTIFY OPTIMUM LEVELS OF NITROGEN FERTILIZATION FOR IRRIGATED CROPS[1]

T. L. Jackson[2]

ABSTRACT

The nitrogen available for plant growth comes primarily from 1) the nitrogen released from soil humus and crop residues, 2) nitrogen added as commercial fertilizers and 3) residual inorganic nitrogen from previous growing seasons or previous crops.

Irrigation insures adequate moisture and relatively uniform yields from year to year. Irrigation also makes feasible the application of fertilizer nitrogen during the growth of the crop and this provides the opportunity to use soil and plant analysis early in the crop season to assess nitrogen needs.

Examples of the use of diagnostic techniques for estimating the fertilizer nitrogen required to supplement residual nitrogen levels and the capacity of the soil to release nitrogen are presented.

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INTRODUCTION

The nitrogen available for plant growth comes from three major sources. Nitrogen is released from soil humus during the growing season. Soil humus, which contains about 5% nitrogen is the major storehouse for nitrogen in the soil. On the average, about 2.5% of this nitrogen is mineralized and released for growth of the crop each year. To estimate the amount of nitrogen that might be mineralized from soil humus we assume that a ha of land to a depth of 30 cm has 3,850,000 kg of soil. If the humus contains 5% of nitrogen the quantity of nitrogen for each 1% humus is 1925 kg. With 2.5% of the nitrogen released by mineralization, the crop would be supplied with about 48 kg/ha/yr for each 1% humus in the surface foot (30 cm of soil).

Nitrogen can be released in the decomposition of crop residues. This release can be significant if these residues contain relatively high nitrogen contents such as might be found when leguminous crop residues are incorporated into the soil.

Second, there can be residual nitrogen present in the soil that might have accumulated from 1) mineralization of humus, 2) decomposition of crop residues, and 3) applied as commercial fertilizer to a previous crop. Nitrogen is also supplied to crops as commercial fertilizer that might be applied before the crop is planted and/or during the early to the middle of the growing season. In some areas the nitrogen in animal manures and in irrigation waters can be important sources.

The residual soil nitrogen is really a combination of all inorganic sources that might have entered the soil or mineralized from the soil. It is residual in the sense that it is there at the end of the previous season or at the beginning of a new season. However, this is the nitrogen we measure in most soil testing programs. It is the "left over" nitrogen that has a potential for contaminating groundwater, and is the "soil nitrogen" that should be considered in modifying our fertilizer recommendations and applications from year to year.

Most soils in the irrigated areas of the western United States were developed under arid or semi-arid conditions. With limited vegetative growth and high summer temperatures to speed up decomposition of crop residues, the levels of soil humus are low when compared with soils in the corn belt and other areas of the United States. The major exception to this would be the irrigated soils west of the Cascades in Oregon and Washington. These low levels of soil organic matter mean that limited amounts of nitrogen are released from humus mineralization during the growing season.

Soils where legumes have been grown in association with an efficient strain of nitrogen fixing rhizobium would be expected to have a relatively high nitrogen release rate but this nitrogen released would come largely from crop residues or decaying nodules rather than from decomposition of humus. In some situations a cover crop is planted in the fall on land that has been heavily fertilized for intensive row crop production. The cover crop helps to control winter erosion and takes up nutrients left from fertilization of the previous crop. If this cover crop is plowed down or incorporated into the soil in an immature stage of growth, decomposition is rapid with release of nitrogen from the incorporated material.

This release of nitrogen from leguminous crop residues, and from succulent (immature) cover crops, is difficult to measure. Dahnke and Vasey (1973) reviewed procedures that require incubating a soil sample at specified temperature and moisture conditions for a two or three week period. Although these procedures provide a measurement of nitrogen released, the time required to ship the sample to a laboratory, complete the incubation procedure, measure the released nitrogen, and send the results back to the grower, has limited the use of this technique as a basis for modifying application rates for nitrogen fertilizer.

Yields are high with irrigation and are generally quite consistent from year to year. This consistency with irrigation helps to establish optimum levels of nitrogen fertilization for a crop for different areas of the western United States. Since many crops, especially cereals and grasses, are fairly efficient in using nitrogen it is possible to combine uptake data with field experiments to arrive at reasonably accurate estimates of the nitrogen required for optimum yield.

Irrigated agriculture offers maximum flexibility in timing of nitrogen applications. This flexibility can be used to achieve an optimum level of nitrogen fertilization. Irrigation provides water in the surface of the soil to help move nitrogen into the root zone and insures that both water and nutrients are available. Sprinkler irrigation offers the alternative of applying nitrogen through the water with the possibility of some uptake through the foliage, especially with urea nitrogen, and maximum movement of nitrogen into the root zone.

These alternatives make possible the use of a combination of both plant and soil analyses during the growing season to measure the supply of nitrogen available to the crop. Fertilization schedules can be developed to supplement preplant or planting time applications plus the soils supply of available nitrogen with applications of nitrogen during the growing season. The level of nitrogen in the plant is then a result of residual nitrogen, fertilizer application, and nitrogen released from crop residues or organic matter.

APPLICATIONS OF DIAGNOSTIC TECHNIQUES

Examples are presented of the use of nitrate nitrogen analyses of soils and plant samples (generally petioles) to determine the status of nitrogen in the crop and thus to estimate the specific amount of nitrogen to add to achieve optimum yield. Applying a significant amount of the crop's nitrogen requirements during the growing season reduces the possibility of leaching losses that can occur early in the season before the root system becomes established.

Sugar Beets

Any discussion of using nitrate nitrogen analyses of petiole samples during the growing season to adjust applications of nitrogen should probably start with the early outstanding work of Ulrich (1948, 1952) with sugar beets. Ulrich and Hills reviewed (1973) information on use of plant analyses as an aid in fertilizing sugar crops. The critical nitrate nitrogen curve taken from Ulrich's work illustrates two important points: 1) the nitrate nitrogen analysis of sugar beet petioles can be related to an optimum supply of nitrogen for this crop, and 2) there is a marked decline in the critical level of nitrate nitrogen as the crop progresses towards maturity. These effects are more important for sugar beets than for many crops because an excess amount of nitrogen late in the season will stimulate vegetative growth at the expense of sugar production.

Carter, Westermann and others (Carter et al, 1971; Carter et al, 1974; Carter et al, 1975; Carter et al, 1976; Stanford et al, 1977) refined the information for sugar beets for use in southern Idaho by establishing information on the rate at which petiole nitrate levels declines during the crop season. A moderate level of nitrate in the petiole that is being supplied with subsoil nitrogen or nitrogen being released from organic sources has the potential of maintaining excess available nitrogen through to maturity of the crop. Roots of sugar beets reaching subsoil nitrogen could account for the increase in petiole nitrate nitrogen in late summer as shown in Ulrich's work.

James (1970) and James and Johnson (1973) identified a range of nitrate nitrogen levels in the soil profile before planting that could be related to the quantity of nitrogen fertilizer required to produce optimum yield and sugar production in central Washington and Utah. A number of fields were found that had more nitrogen in the profile at planting than was needed for optimum sugar production; added nitrogen on these fields decreased sugar yields.

County agents and other agronomists (personal communications) have found 450 to 680 kg residual inorganic nitrogen/ha in the 0- to 75-cm depth of soil in a number of luxuriantly fertilized fields in central Washington and the Treasure Valley area of southern Idaho and eastern Oregon.

Another complicating factor has been identified by Nielsen and Banks (1960) in Utah where most of the sugar beets are grown under rill or furrow irrigation. Water applied to a furrow which is sometimes 30 cm lower than the top of the adjacent mid-row where the beets are growing, moved by capillary action into the middle and top of the row. This water carries nitrate nitrogen with it and often results in a crust of nitrate nitrogen and other salts at the highest point of the rill ridge by mid August. Late summer thunder storms or rains can move this nitrogen down to the root zone where it acts as a late summer application of nitrogen that is difficult to control. Obviously this would not be a problem with sprinkler irrigation or in areas where rains are more frequent throughout the growing season.

Winter Wheat

The use of nitrate nitrogen in stem tissue of small grains to predict needs for added fertilizer has been investigated by Gardner et al (1976) and Gardner and Jackson (1976) at Yuma for barley and by Brown and Jones (1977) for winter wheat in southern Idaho. In these examples, the nitrate nitrogen concentration in the second and third internodes at tillering or an earlier stage of growth can be used as a basis for predicting the nitrogen needed by the crop. This is a very early stage of growth to evaluate the potential supply of nitrogen for the season, especially in the work by Brown and Jones in Idaho (1977). Three important factors contributed to this.

1. The root system of winter wheat develops in the fall and continues to grow through part of the winter. By the time spring growth starts, a significant amount of soil depth has been exploited by the root system since many of these soils are relatively shallow (less than 3 feet or 1 meter).
2. Most of the crop available nitrogen in these soils at the end of a growing season will be in the nitrate form that is readily taken up by the crop since soil organic matter levels are relatively low. Thus, residual nitrogen left from potatoes, onions, sugar beets, sweet corn and other intensive row crops will influence the nitrogen content of the following crops at an early stage of growth.
3. Nitrogen fertilizer can be added to a cereal crop any time before jointing, if surface moisture is available to move the nitrogen into the root zone, without reducing the efficiency of use or the yield increase from the applied nitrogen.

Potatoes

Potatoes have somewhat different nitrogen requirements than many row crops since they have an indeterminate habit of growth and can be harvested as an early crop 90 or 100 days after emergence or after an extended growing season of 6 months. Lorenz et al (1964) and Geraldson et al (1973) developed a relationship between nitrate nitrogen of petiole samples and optimum yield of potatoes where harvest followed a 3 to 4 month growing season.

Potatoes in the Columbia Basin of Oregon and Washington have a wide range of growing seasons. Planting starts the last week of February on many years in the lower elevations around Hermiston, Oregon and Pasco, Washington. Russet Burbank potatoes planted as early as March 1 are sometimes maintained in vegetative growth through

October. High rates of nitrogen, 340 to 450 kg/ha, are required to maintain a crop in a vegetative state of growth for this period of time and to produce the yields (80 to 90 metric tons of potatoes/ha) that have been achieved with center pivot irrigation on sandy soils. The shallow root system of potatoes makes it advantageous to use growing season applications of nitrogen on the sandy soils in the Columbia Basin, Pasco-Hermiston areas, where these practices are followed.

Consulting agronomists in this area emphasize maintaining a specified level of nitrate nitrogen in petiole samples using applications of soluble nitrogen fertilizer through sprinkler irrigation systems. Optimum levels of nitrate nitrogen in petiole samples for specified dates before harvest have not been established with reliable field research in this area. It is generally accepted that more than adequate levels of fertilizers are being applied for production of this crop. The nitrogen required to produce a crop of wheat is frequently supplied as residual nitrogen after the potato crop is harvested. This level of fertilization used undoubtedly results in nitrogen being leached below the rooting depth of potatoes on sandy soils.

Tree Crops

Tree fruits present a distinctly different problem in terms of nitrogen management (Kenworthy, 1973). Trees are a perennial crop with the same plant occupying an area for many years. This offers the opportunity to evaluate the previous year's program and make adjustments on the following year's nitrogen application. Also, excessive nitrogen can delay ripening and development of color and sugar for many fruit crops so that crop quality can be reduced with excessive nitrogen fertilization.

Field experiments have shown that many growers were using excessive rates of nitrogen for citrus production in southern California and that these high rates of nitrogen reduced the quality of fruit produced (Embleton et al, 1974; Jones et al, 1968; Jones et al, 1970; Jones and Embleton, 1967). Leaf analyses from these experiments have identified critical levels that can be used to monitor nitrogen fertilizer programs to achieve optimum yield and quality (Embleton et al, 1975). Excessive use of nitrogen resulted in leaching losses and nitrate nitrogen in the drainage water flowing from some of these orchards (Bingham et al, 1971; Embleton et al, 1974; Embleton and Jones, 1974). The results from this research have been used to modify fertilizer programs to reduce fertilizer costs, improve quality of fruit produced, and limit movement of nitrogen below the root zone of the crop.

I have discussed four examples where plant and/or soil analyses for nitrate nitrogen have been used to predict the optimum rates of nitrogen fertilizers to be applied for crop production under irrigation. Many other examples could be given.

Farm advisors, county agents and other field men are familiar with specific calibration data that can be used to modify nitrogen fertilizer application in local areas throughout the west.

A summary of this information available for California has been published in Extension Bulletin 1979 (Reisenauer, 1976).

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ECONOMIC CONCEPTS AND POLICIES RELATED TO CONTROLLING NON-POINT SOURCE POLLUTION STEMMING FROM AGRICULTURE

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ABSTRACT

This paper describes the role of economics in solving non-point source pollution problems in terms that are understandable for the noneconomist. An economist attempts to watch over the process of converting natural resources into marketable commodities to assure that the process achieves the greatest possible good for members of society. Economic tools are well equipped to handle this process for goods and resources known as private property, those items used exclusively by the individual owners. Unfortunately, most pollution problems occur through the use and misuse of public property, those items owned and used jointly by all members of society. Air and water are the most common examples of public property.

Economic theory of a firm or factory assumes that society uses well defined signals to tell the producer how much of any product to produce. Market prices are the signals sent from the consumer to the producer. The system is imperfect, however. There are residuals of nitrates, smoke, dust, and noise which are external results or by-products of activities carried out in normal production processes. These residuals or "externalities" are not reflected in the prices derived by our market system. The result is often to over-produce both consumer goods and pollution. The paper describes economic concepts and public policies that are useful for internalizing the costs

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caused by pollution and abating or solving pollution caused problems.

The concepts of opportunity cost and income distribution and their role in pollution abatement are discussed. Methods of choosing levels of abatement include the extremes of zero pollution and that which can be achieved by best management practices. The economic and social optimum probably lies between these extremes. Arbitrary standards are sometimes imposed as a compromise between the extremes of zero pollution and no abatement in an effort to approximate optimal levels of abatement.

INTRODUCTION

Economic activity is the process of turning nature into products that satisfy human needs and wants. Sometimes this is done directly. Wind is harnessed to pump water for an isolated farmstand. Other times, this is done in a roundabout way. Ore is mined, smelted, and cast into pigs. The pigs are shipped, melted, stamped, rolled, shaped, and eventually become an automobile. Although the trip from the Mesabi range to Detroit and then to the local dealer is very complex, each step requires that at least some of nature become a part of man's activities. It becomes part of the production, exchange, and consumption that characterizes a modern industrial society.

This would be only mildly curious except for two things. First, nature is finite, but man's wants seem not to be. Man makes a claim on nature. This claim enables water or rock or timber to be turned into economic goods, but in most cases the transformation of nature into economic goods is an irreversible, one time process. Once the first transformation has been made, the total available stock of natural endowments has been reduced. One man's incursion on natural resources limits the activities of all other persons because resources are limited in supply. A second problem arises in connection with those few natural endowments that appear to be free. Air, sunlight, moving water, and some public lands are free to all who want to use them. They acquire this cost-free characteristic because they are ridiculously plentiful or because it would require unmanageable expense to harness these resources and make the users pay for them. The fact that they are free has made these resources subject to tremendous abuse. Smoke is emitted into (free) air; sewage is dumped into (free) rivers and lakes; hot dog wrappers are thrown on the (free) grass in the public park.

Smoke, sewage and hot dog wrappers went largely unnoticed until fairly recent times. True, in the late 14th century, King Edward III prohibited dumping garbage into the Thames because those wastes prevented ships from coming up the river to the city of London. This was, however, an unusual event and Edward III stood alone among monarchs. The removal of smoke, sewage, and hot dog wrappers is a very recent phenomenon. Until (perhaps) 1960, man lived so close to nature that to try to clean up the mess was a luxury he could ill afford. Since then, the view has changed. Economic activity always impinges upon and often impairs the natural world. If society is to protect the natural world, it must actively intercede on behalf of the unutilized, underutilized, and endangered resources.

Agriculture is a smug but difficult industry when cast in this milieu. It is smug because it knows that its prime task is to turn nature into meat, vegetables, and fruit. It uses resources but is generally proud of its record of "wise use" and its infrequent incursions into the domain of others. Feedlots have smelled and have had their wrists slapped, but the individual farmer has not worried about polluting anything or bothering anybody.

All this is changing. It is now known that many fertilizers and pesticides and other items applied to agricultural land can slip into surface or underground watercourses and cause damage to the water and its uses. It is known that small - even miniscule - amounts of soil erosion can bring sediment damage, allergies, or flooding hundreds of miles away. Unlike industrial enterprises which can be pointed to and told to mend their ways, agriculture produces unidentifiable sources of pollutants that have come to be called non-source point pollution. Developing the means to deal with any type of pollution is difficult; dealing with the non-point variety is almost impossible. This paper is designed to show why this is so. After a brief opening comment regarding the task of the economist, the paper turns to property rights, some lessons in economics, some hypothesized methods for pollution abatement, and a plan to use in attacking non-point source problems.

THE ROLE OF THE ECONOMIST

The primary concern of the economist is the allocation of scarce resources among competing uses. An economist attempts to watch over the process of converting nature into commodities and to make sure that the process achieves the greatest possible good. This "greatest good" is related to efficiency and to the distribution of economic rewards among members of the community. The tools of economic analysis are especially well equipped to study and make policies related to the class of goods or resources known as private property. Private property - goods or resources - can be traded and take on value (prices) that reflect their relative worth in creating welfare. There is, however, a large class of resources that are not susceptible to restrictive property rules and are known as common property resources, or, simply, "community property". Common property includes those resources that are not exclusively controlled or owned by an individual and, hence, can be used by everyone.

It is often remarked that "everyone's property is no one's property" and this phrase illustrates the economist's concern when confronted with the many problems of the environment. If common property is present, everyone will use it; no one will pay for it, and no one will take the trouble to care for it. Environmental and ecological problems arise in such a setting. One individual, in striving to use common property to achieve his objectives, may cause detrimental effects to be imposed on others. Such a phenomenon is known as an external diseconomy. While society is generally well equipped to handle the private uses of private property, it has not developed adequate rules for using common property or for handling the external diseconomies that arise from such use. Understanding this problem requires a thorough investigation of property rights.

PROPERTY RIGHTS

Property rights are the rules that say who can do what with land, capital fixtures on the land, and nearly all classes of natural resources[2]. These rights are established by society and at least partially determine the extent of environmental problems that will surround a particular class of resources. They also provide many clues about the means of solving problems of environmental degradation. Unfortunately, the rules by which we operate today were designed in a period in which there was an abundance of common property and very little concern about how one person's use of a resource would affect the value of that resource to others. As time has passed, pressure on resources has increased and the earlier laws have become obsolete. This is especially true in cases in which the distinction between common property and private property has become blurred. Actually, these are merely the extremes on a continuum ranging from the most mobile, exclusive private property item (such as a toothbrush) to those resources held in common by all people in an economic society - air, radio signals, and Yellowstone National Park.

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[2] Property rights also apply to non-land items such as automobiles, furniture, and the right to work. The discussion here is limited to that class of resources that is found in nature.

Private Property

The private property rights conferred upon the owner of a toothbrush provide him with the privilege of exclusive use and the right to transfer the use of the toothbrush to anyone else whenever the transfer becomes advantageous. In other words, an individual's right to an item can be traded to someone else for an agreed upon value. Generally, the value of the privately held item is determined in the market place. Society may impose rules that restrict the use of the item, but the individual who holds title to the item is the only one who can use the item even within the bounds of the law. For example, a shotgun is private property and (usually) one person has exclusive rights to its use. Society has, however, imposed many rules that restrict use of that gun. In general, the restrictions are designed to ensure the safety of those who are near the gun owner. Since safety has different meanings in different places, the rules for the use of this private property item may be changed over time or space by the will of society.

Economic theory is extremely useful for explaining the values and the methods of exchange for private property items. If all activities of all people were carried on exclusively through private property, the number and magnitude of environmental problems would be greatly diminished. The market would measure the positive value (utility) derived from the use of that good and it would reflect the damage to others caused by one person's use of the same good. Unfortunately, the market is not up to this task and a great deal of disutility (loss of value) is created by the use of private property items such as automobiles when they emit exhaust fumes or fertilizer when residues from the fertilizer are carried into someone else's property by return flows.

Common Property

Common property rights appear at the opposite end of the spectrum from private property rights. They are not owned exclusively by one individual and are therefore nontransferrable. Truly common rights are vested equally in each citizen, and allocation of common property resources among individuals becomes problematic. Many common property rights are allocated on a simple availability or first come-first served basis. Two general types of services flow from common property rights. One set is subject to limited individual choice, while the remaining set of services cannot be selected or controlled by the individual. An example of the former would be sailing, water skiing, or fishing services available from a publicly owned lake. Individuals may choose to utilize the wind for sailing or the water for fishing or skiing. If they are rational, the choosers will participate in the activity up to the point where their marginal gain in satisfactions equals the marginal costs of participating in the activity. Climatic conditions are an example of common property resources which are not subject to individual choice. When it snows on me, it snows on my neighbor, too. Or, to use a technologically induced example, both increased rain and increased sunshine may occur as a result of seeding clouds. But no person within the affected area can choose how much of each he will take. He must accept the same rainfall and sunshine as everyone else.

There is no market where users of common property resources can interact and negotiate to their mutual advantage. The relative goodness or badness of this resource use manifests itself in ways other than money prices. If one person's use of a common property resource serves to damage the incentive or reduce the profits of a second person, the second person has absorbed an external diseconomy and this will be reflected in the value of his other resources. The familiar kinds of pollution yield this effect. If one person's use of a common property resource serves to increase the incentive or income of a second, the second person will have absorbed an external economy. An external economy occurs when one farmer's drain system also lowers the water table on his neighbor's land.

If the rights to common property resources were rationed or sold in orderly markets, the affected parties would bargain with one another over how much of each should be used and what kinds of activities they should conduct. Since no such market exists, external effects abound and add to the problem of environmental degradation. Environmental degradation, or pollution, occurs when one user of a common property resource neglects to consider the fact that others may also wish to use the resource. Since there is no price for the user to pay, he uses as much as he pleases without regard for others. The market fails and the economist as well as the planner is left with few rules on what can or should be done. The whole concept of resource allocation loses its meaning and no one can say unequivocally that too much or too little is being used or that too much or too little is being produced.

ECONOMIC CONCEPTS

Economists bring a whole set of tools into the study of the above problems. The classical theory of the firm is probably the most prominant. The theory of the firm assumes that society uses well defined signals to tell the producer how much of any product should be produced. Prices serve to guide resources into their most beneficial uses and the prices of factors or production and goods accurately represent the contribution each makes to social welfare. If knowledge is perfect, even the level of pollution will not exceed society's desires. Total welfare is optimized when each producer's profits and each individual's satisfactions are maximized.

This hypothetical model presumes that either all inputs are fully converted into outputs with no residuals created or that the environment's capacity to assimilate such residuals is unlimited. Alternatively, the model assumes that every price automatically takes into account the effects of residuals or pollutants. The homemaker who buys beef in Fresno is assumed to be paying for the damage caused by odors near the meat packing plant in Omaha. Obviously, those hypothetical conditions do not hold in the real world. Residuals are not completely absorbed and homemakers are not fully compensated.

Externalities

An economist describes these imperfections as "externalities". Residuals, pollution, smoke, and noise are external results or by-products of activities carried out by an individual or firm in the process of maximizing his welfare or its profit. A farmer will strive to maximize profits by combining land, water, fertilizer, labor and sunshine to produce crops in an optimal and efficient way. In the process the farmer may create externalities such as air pollution (dust or odors) or water pollution (sediment and nutrients) which can harm or reduce the welfare of others, but the farmer's instinct tell him to maximize his own profits without regard for the welfare of his neighbors. He presumes that maximizing his own welfare will help to maximize the welfare of society and since all the neighbors harbor similar thoughts, each attempts to maximize profits and each disregards the damage he, himself, is imposing on others.

Externalities or pollutants are not all alike. They may be classified as either pecuniary (priced or having market defined value) or technical (nonpriced). Pecuniary externalities are those that occur when the actions of some individuals or firms result in price changes for others. For example, when Eugene, Oregon, drops raw sewage into the Willamette River, it imposes pecuniary externality on landowners who live downstream. Technical externalities are of a different kind. They affect the welfare of others, but not in an economically measurable sense. Developing a very noisy airport may preclude the installation of a school or a hospital on surrounding territory. The airport creates a technical barrier but its effects are not measurable in economic terms.

Opportunity Cost

Because many externalities are not easily measurable, it is necessary to devise elaborate schemes to determine their impacts. The economist relies heavily on the concept of opportunity cost for this purpose. Briefly defined, the opportunity cost of a resource is its value in its best alternative use. Water used for irrigation may yield a profit of $15 per acre foot. If the same water were diverted to generate power, it might have a value of $10 per acre foot. This $10 becomes the opportunity cost of water used for agricultual production.

This concept can be used to assess the impact of pollution that reduces the recreational value of a lake. In this case, the value of the lost recreation is measured by the costs required to develop alternative means of recreation. Construction and maintenance of swimming pools, travel costs to more distant lakes for boating, fishing, or water skiing may be involved[3/].

The opportunity costs of resource uses affected through externalities can be used to measure pollution costs and to help determine whether or not pollution abatement programs should be undertaken. Although the opportunity cost is the key to damage assessment, the distribution of externality damages among members of society is also useful in rendering opinions about allocating the cost of pollution abatement among society's members. In a sense, the size and distribution of opportunity costs become substitutes for the true economic

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[3/] Values may also accrue to the lost recreational opportunities that are irreplaceable or unaffordable through conventional alternate means. The sum of these values becomes a part of the opportunity cost of the lost recreation, but they defy actual measurement at this time.

values of pollution costs and they become a standard against which the benefits of pollution abatement can be judged.

Costs can be described with respect to their magnitude, their distribution, and who bears them. The total costs of pollution can be high or low, but the real test comes in deciding who is responsible for them. If my chickens get out and ruin your garden, the question of distribution of costs is easily settled. If the return flow from irrigation in the Upper Colorado River drainage causes damages throughout the desert southwest, the matter becomes more complex.

Income Distribution

In general, costs occurring because of my chickens and your garden can be measured as the change in income of welfare that you and I receive. Your income is reduced; mine is increased (because I do not have to buy feed for the chickens). In some cases, it is possible to add the changes in income of individuals to derive an estimate of total costs and benefits that stem from a particular activity including such activities as polluting or pollution abatement. In other cases, the costs may be so dispersed that no private entities are susceptible to significant change. In those cases we must introduce some arm of government and ask it to become involved in setting and enforcing standards, collecting taxes, redistributing wealth; and carrying out other activities that are beyond the scope or management of individuals.

We are not here to discuss my chickens in your garden, but, rather, the more complicated case occurring when many pollutors each contribute an unknown amount of pollutants that eventually affect an unspecified number of people. This is not an easy task, but one thing is certain: No private entity has enough knowledge, enough power, or enough incentive to undertake the task. If the problem of non-point source pollution is to be solved, the solution must come through group activity sponsored by the government. The example of farmers in the Upper

Colorado River Basin provides a vehicle for discussing several aspects of the problem.

Assume for a moment that farmers in a river basin are polluting the river because their irrigation return flows carry residuals from fertilizers that have been applied to their lands. The government may seek to reduce pollution by imposing a tax on either the fertilizer, the water, or both. As a result of the effective increase in the price of these inputs, farmers will use less of each and in so doing will reduce the volume of pollutants reaching the river. This reduction in pollutants will help meet our objective since now the river is "cleaner". The cleaner river does not, however, come without cost. Several things follows from the individual farmer's decision. Reducing the use of inputs will reduce the amount of product and this in turn will reduce farmers' net incomes. The aggregate reduction of net income for all farmers in the basin becomes one measure of the costs of pollution abatement.

In the process of reducing pollution, however, society has collected a new tax from the farmers who continue to apply fertilizer and water. Revenues from this tax can be redistributed to other members of society to help compensate for the reduction in water quality. Thus the net (or total) effect of a taxing program becomes the sum of lost net farm income minus the new tax revenues. A problem still remains, however, because no one can be sure just how the tax rates should be set or just how the tax revenues should be distributed.

The government can take a second approach to the problem. It can choose to pay (bribe) farmers not to pollute the river. The bribe induces the farmer to reduce the use of pollution inputs, but it maintains farm incomes by transferring general tax revenue to the farmers. The direction of income transfers in this case is opposite from the flow in the case of a tax program. Which is better? No one can say for sure.

A third method of control calls for the government to impose a legal limit on the quantity of offending materials that can be used. This limit would be enforced through threat of fines or jail sentences, and there would be no tax or transfer of payments involved. Since farmers would still have to comply with the rules, they would experience a reduction in net farm income. The river would be clean, but one group would be made economically worse off, while some elements of society would benefit from a cleaner environment. Unfortunately, in none of these cases can we be sure that benefits exceed costs.

It now becomes clear that the means of controlling pollution - especially the capricious non-point source pollution - should be chosen with some care. The choice should be tempered by some careful reflection regarding the distribution of the costs and benefits accruing to each of the alternative plans. The effectiveness of a pollution control program may appear vastly different depending on the vantage point. A tax penalizes the pollutor and provides revenue. A bribe asks society to buy out the offender. A law reduces the income of the offender and the amount of product available to society. It is not clear which means of control is most desirable in non-point source cases.

DETERMINING THE "CORRECT AMOUNT" OF POLLUTION CONTROL

Most U.S. citizens will agree that the territory that composes the United States is now much too heavily polluted. Most would similarly agree that something should be done about the problem. There is little agreement, however, on how much should be done because no one knows the magnitude or distribution of costs and benefits associated with each level of pollution or each level of control. One thing is certain: Each of the several methods of control will, if left to itself, yield a different amount of control. A tax may be 60% effective in removing pollutants (and 100% effective in reducing the pollutor's incomes!). A bribe may be 40% effective and a law can be as effective as the administrative unit that makes the law and the policy unit that enforces it. Interested and affected parties in a given non-point source pollution case will probably choose different criteria depending on whether they are on the side of the polluters or on the side of those who are injured by the pollution. The decision is thus arbitrary. This section will describe some of the more evident means of setting abatement levels or environmental standards for agricultural pollution problems[4].

Zero Pollution

Zero pollution is not a means, but an end. Zero pollution is mentioned here because of the large outpourings of interest surrounding the nation's capacity to rid itself of all forms of pollution. Zero pollution is an extreme position advocated by a small number of zealous individuals and a smaller number of environmental groups. It is questionable whether zero pollution - as conventionally defined - can be met in nature let alone under the conditions imposed by an industrial society. Indeed, air quality standards for western Colorado were recently set so high that they were occasionally violated by natural conditions. Rules insisting on zero pollution are not generally taken seriously because most people realize that there is little room for economic activity in such cases. We know of no serious conversations that could lead to a zero pollution standard for that segment of agriculture that produces non-point source pollution.

The Economic Optimum

Common sense indicates that there can be too little or too much pollution. If these two extremes can be identified, there must also be some amount that is "just right" and that amount is the economic optimum. In a perfect world where all effects and all costs are measurable, the optimum is defined by extending pollution control activities to the point where the added dollars of expenditures or control are just offset by the added dollars in benefits. This is a familiar rule in all of economics and it has a common sense interpretation: Keep doing something as long as it pays.

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[4] The standards to be met by non-point source polluters in agriculture have not yet been determined. Because of this, one can find a wide range of views regarding the "appropriate" standards which should be set. At this point, no particular view can be judged to be either right or wrong.

One can imagine a large industrial plant dumping waste into a river. The water is subsequently taken from the river to be used for a municipal water supply. Pollution abatement procedures could be initiated to force the plant to clean up its own mess. The plant would pay and the downstream city would benefit. As long as increased costs of abatement were covered by dollar savings to the city, society could recommend that more and more pollution abatement capacity be added to the plant - marginal costs would be exceeded by marginal benefits.

The method of approach, however, breaks down in a non-point source pollution case. Although it is known that an optimum exists, the physical and economic data appropriate to that optimum are largely unknown. It is not known whether one of the hundreds of polluting farmers could solve the whole problem if he were to discontinue all activities that contribute to the problem. Nor is it known whether or not a 10% reduction in all polluting activities would reduce pollution by 10%. Because of this lack of knowledge, the economic optimum cannot be precisely determined. All that is known is that it is more than zero; less than present levels. Policy makers are forced to choose one of two approaches to the problem. They can recommend that all farmers use the best possible management practices to achieve an acceptable level of pollution control or they can set arbitrary standards and attempt to enforce these by law, taxes, bribes, or combinations of these.

Best Management Practices

Those working closely with agriculture often claim that the only realistic pollution standards for the agricultural industry are those that can be met by applying the best available management practices to each farming enterprise. This is a peculiar position to take. It requires that farmers utilize the best method of combining inputs to yield a profit maximizing output. It says nothing whatsoever about the relationship between economic activity and pollution. This is a "leave-things-alone" position that suffers from over-simplification. There are some problems with this kind of reasoning. Recommending best management practices (BMPs) always begs a question: Best with respect to what? There are BMP schemes that relate to water, to land, to the farmer's labor, or to the farmer's credit rating. They are not necessarily the same and, since they are not the same, each best practice will yield a different pollution loading for a stream or for the atmosphere.

It is likely that in most cases the application of BMPs would improve water quality or environmental conditions affected by agriculture. In the fertilizer/return flow case, BMPs might require the farmer to adopt a method of water application that resulted in lower return flows. These lower return flows would carry smaller amounts of fertilizer residuals back to the stream so the stream would be cleaner; the increased parity coming as a by-product of improved resource management. Although BMPs can be recommended on efficiency grounds, there is no guarantee that these practices would produce a socially desirable amount of pollution control. There also remains the problem of how to get farmers to adopt a specific set of management practices, particularly if the practices increase production costs or reduce income. Otherwise, one must assume that the "good" farmers are not non-polluting and that others should merely follow in their footsteps, a position that is difficult to justify.

Arbitrary Standards

Although the word "arbitrary" has a terribly unscientific and nontechnical ring, it is often the only way to approach pollution abatement.

It will remain so until data systems are improved. But "arbitrary" is indeed tempered by reason so arbitrary rules are not often to be confused with ridiculous rules. Agencies charged with the responsibility for improving the environment are frequently left with few rational criteria for setting standards. They must strike a balance between affected parties demanding complete abatement and polluters who provide convincing arguments for continuing current activities. The result is often a bounded set of arbitrary standards that may be designed to prevent further degradation of a local environment or designed to meet certain health related criteria for water or air quality. The final set of standards will probably depend upon some combination of the attitude of the enforcement agency, the relative bargaining power of affected parties, the apparent seriousness of the pollution problem, and the knowledge and tools available for alleviating the problem.

Using arbitrary standards brings some disadvantages, but has some merits, too. Since conditions are likely to vary considerably from one geographic area to another, a general, or nationwide, standard would not solve many problems. It would be too strict for some areas; too lenient for others. This is, of course, a disadvantage since each area must develop its own standard and that requires time, effort, and money. It also carries the distinct advantage of requiring each affected area to study carefully its own problems and this necessary element of local citizen input is quite likely to make the eventual resolution more palatable.

Even using arbitrary standards for pollution abatement, there is still the problem of how best to achieve the chosen standard. The choice of the abatement enforcement tool will ultimately determine the costs of abatement and the distribution of those costs among society's members.

DISCUSSION

The problem of pollution has become an important fact of the U.S. political economy. It is a serious technical problem and a serious economic problem. It has become a political problem and an emotional problem for some individuals and some groups. There is little doubt that this industrial society has the technical means to create a pollution-free environment or, at least, a tolerable environment. The question remains: Do we want to? That question is not easily answered.

Most pollution problems arise because of ambiguous rights in property or market failures. An industrial plant dumps smoke into the atmosphere because no one has "rights" to the air so no one can order the plant to quit using that resource. A homeowner can irk his neighbors by mowing his lawn at seven o'clock on Sunday morning. He can continue to do this because there is no organized market in which the neighbor can buy silence - the market for this service has failed to develop.

An industrial polluter or a residential lawn owner can be controlled through fairly straightforward means. Zoning regulations, pollution control devices, laws, economic incentives, and peer pressure. The problem becomes extremely difficult when the source of the pollution is not easily recognizable and when those who must endure the lower quality environment are a public group who may have nothing in common except the fact that they are imposed upon by the same pollutants coming from the same source. Although traffic noise and auto emissions in a metropolitan area have some characteristics of non-point pollution, attention is now focussing on agriculture.

U.S. agriculture is a highly intensive industry that uses complex sets of inputs to produce huge volumes of foodstuffs. The industry, especially as it is organized in the West, has been held up as a picture of efficiency and a model for other parts of the nation and the world. Only recently have environmentalists begun to point to agriculture to indicate that some environmental problems may arise in that sector of the economy. Insecticides and herbicides may be applied in such a way as to produce wind-born residuals that affect cities many miles away. Fertilizers and other inputs applied to irrigated land may be leached into streams and eventually pose a threat to aquatic life or produce new forms of equatic life that interfere with the traditional uses of the river or stream. This type problem is insidious and, although it is justifiably being noticed, the solutions to the problem are often equally insidious. This non-point source pollution can be ended by stopping agricultural production or at least reducing output to pitiably small levels. It may be stopped by generating a new class of inputs that leave fewer residuals or are less amenable to transport into other areas or into common property resources. In any event, the structure of the industry will change in response to rules about input use. The new rules will also change the output mix of agriculture, the size and distribution of income within the industry, and the relationship between agriculture and the local and national economies.

Many people view these kinds of problems as technical. That may be true, but the economist has a stake in these things, too. The economist's specialities include studies of cost, efficiency, demand, and income distribution. An economist can make refined recommendations about how much fertilizer should be applied, how much hay should be fed, and whether or not the cows should be milked by hand or by machine. This is so because each of these activities centers on the use of exclusive, private property and the technical behavior as well as the market behavior of each of these inputs is well known.

The economist is not so well equipped when it comes to studies of non-point source pollution. By definition, fingers cannot be pointed and blame cannot be assigned. The production of the pollution is not well understood and its effects can be only estimated. Nonetheless, the economist should be invited to work with both the technical expert and the policy maker. The economist can tell the technical expert something about the costs of each activity and he can make judgments regarding how these costs might be distributed among suspected polluters or beneficiaries of a clean-up program.

The best work of the economist will come from his alliance with the policy maker. Previous sections of this paper indicate that there is a certain amount of arbitrariness associated with pollution control policy. Regardless of this arbitrary characteristic, the policy will always divide society into gainers and losers. A tax will impose costs on one set of people; a bribe will cause others to bear the cost. An arbitrary allowable level of pollution will cause still another arrangement of distribution of gains and losses. The economist is trained to study these and his expertise should not be overlooked.

In sum, pollution, degradation, and despoilation are problems stemming from technical activities. It is reasonable that most solutions to the problems be sought by investigating technical relationships. Thus, the engineers, the limnologists and ecologists, and the foresters must play important roles in assisting to solve the problem. But one major--perhaps the major--consequence of a despoiled environment is a rearrangement of social relationships existing among individuals and groups. It seems only reasonable to expect the economist to become involved in these problems and their solution.

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A CASE STUDY-
NITRATES IN THE UPPER SANTA ANA RIVER BASIN IN RELATION TO GROUNDWATER POLLUTION

R. S. Ayers[1]

ABSTRACT

In response to a request from the Santa Ana Watershed Planning Agency (a regional planning agency) the Kearney Foundation of Soil Science of the University of California conducted an interdisciplinary 3-month study of the nitrate problem in the Basin. The study was limited to the upper part of the basin where nitrate degradation of waters was more serious.

The study included 1) a review of available data to identify existing areas of high nitrate concentrations in underground waters, 2) a review of past land use, water and fertilizer use and waste disposal practices, and 3) estimates of the impact of irrigation, fertilization and use of animal wastes on leaching of nitrate from root zones. Guidelines for the use of water, fertilizers and manures were developed.

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INTRODUCTION

Nitrogen is present in a number of areas in the underground waters of California. In several instances it exceeds E.P.A. and U.S. Public Health drinking water standards. Two previous reports had discussed the problem of nitrate in groundwaters in specific areas to a certain extent (Stout et al, 1965; California State Department of Water Resources, 1968). Considerable publicity was generated in 1966-67 when two of the nine city wells at Delano, Kern County, California were reported to have nitrate concentrations well in excess of drinking water standards. Following these and other similar reports from scattered areas of the state, interest in nitrates in groundwaters in California increased. From published well water analyses (DWR Bulletin 130-series) a statewide distribution map of nitrates was prepared for use in Cooperative Extension to alert our extension audience to the problem of nitrates in the state's underground waters.

Since agriculture is a large user of nitrogen and nitrogen was known to be present in the groundwater, it seemed to much of the public that this was a clear cut case of agricultural pollution and should be stopped. Clearly those of us working closely with agriculture needed answers. We needed immediate answers to questions that had not previously been asked - at least not in the very loud, demanding voice of 1970-71.

THE SANTA ANA PROJECT

One such area with apparent nitrogen problems in the ungerground waters was the upper Santa Ana River Basin of Southern California. It had a concerned planning agency - the Santa Ana Watershed Planning Agency (SAWPA) - which was charged with planning for complete water resource management. This agency had identified nitrogen as a problem and wanted some sort of evaluation as to its extent, probable cause, and some ideas as to how to correct or adequately manage the nitrates to prevent further degradation. The underground waters are the primary source for water users in the upper basin and waters that pass from the upper basin through the Prado dam become recharge water to the underground, the primary water supply for Orange County and its nearly two million people.

The agency asked for help from the University of California (UC) - Cooperative Extension and Experiment Station - through personal contact with the UC Riverside Extension Soil and Water Specialist. He, in turn, enlisted the help of others and together with SAWPA informally worked out the details of an agreement as to what UC could do to help SAWPA. A formal request was submitted by SAWPA asking the Kearney Foundation of Soil Science (a statewide organization within UC) to make a study on "Nitrates in the Upper Santa Ana River Basin in Relation to Groundwater Pollution".

The Kearney Foundation agreed and a group of interested Extension and Experiment Station staff were invited to participate. No reimbursement from SAWPA was involved except that some unstated "Grant-in-Aid" might be forthcoming at the conclusion of the project. The Kearney Foundation was to sponsor and fund the project, which it agreed to do. The effort was considered to be an interdisciplinary learning process in which UC would bring its resources to bear on a problem for a short time to seek answers based primarily on existing data with only a very limited amount of development of new data.

After an initial meeting the group met about every two weeks over a three-month period to report on progress, ask questions, and assign tasks. The study group included a truly interdisciplinary group. Included were the following from Cooperative Extension: 2 Soil and Water Specialists, 1 Irrigation Specialist, 1 Agricultural Engineer, 1 Agricultural Economist, and 1 Staff Research Associate. From the Experiment Station: 2 Hydrologists, 4 Soil Scientists, 1 Systems Analyst, 1 Sanitary Engineer, 1 Sub-tropical Horticulturist, 2 Olericulturists, 1 Soil Microbiologist, and 1 Water Scientist.

What Did We Do?

1. Nitrogen sinks and sources were identified.

   Various large scale world wide nitrogen sinks and sources were quantified. Then these were narrowed down to the Upper Santa Ana basin and schematically presented as nitrogen pools and fluxes, figure 1.

2. Locations of high nitrate concentrations in groundwater were identified, Figure 2, in which the concentration contours are in mg nitrate/l.

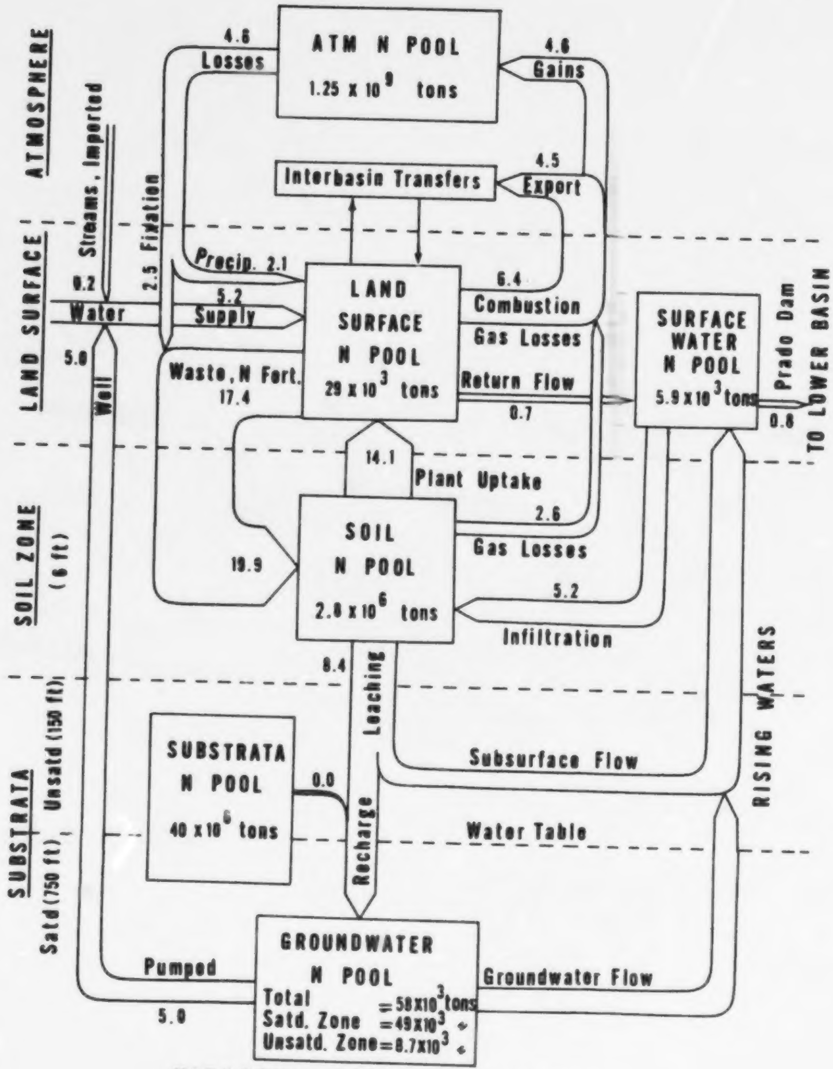
   Water analyses covering the basin were available from the California Department of Water Resources dating back as far as 1919. These were entered on base maps to identify nitrate "hot spots". Changes in nitrate concentrations in both underground and surface streams also were evaluated.

3. History of land and water use and fertilizer rates and waste disposal practices surrounding each of the "hot spot" areas were investigated.

   Crop acreages in some instances had changed over the years but in general land use stayed about the same. Fertilizer rates, however, had changed. Rates for citrus had been greatly reduced - attributed primarily to the adoption of plant tissue analysis as a management tool to improve efficiency. In contrast, fertilization rates for vegetables had increased dramatically.

   Sewage effluent, sludge, and even primary sewage had been disposed of in the immediate vicinity of the two primary nitrate "hot spots". Since this study was to evaluate agricultural inputs, we did not attempt a more detailed study of the effects of sewage disposal.

   One great change in the basin nitrogen loading had been the influx of dairies and dairy animals into the Chino Basin. At the time of the study 130,000 dairy animals were confined within an


ATMOSPHERE
LAND SURFACE
SOIL ZONE (6 ft)
SUBSTRATA
Unsatd (150 ft)
Satd (750 ft)
ATM N POOL
1.25 x 10^9 tons
4.6
Losses
4.6
Gains
Interbasin Transfers
4.5
Export
Streams, Imported
2.5 Fixation
Precip. 2.1
0.2
Water
5.2
Supply
5.0
Well
LAND SURFACE N POOL
29 x 10^3 tons
Waste, N Fert.
17.4
6.4
Combustion
Gas Losses
Return Flow
0.7
SURFACE WATER N POOL
5.9 x 10^3 tons
Prado Dam
0.8
TO LOWER BASIN
14.1
Plant Uptake
2.6
SOIL N POOL
2.8 x 10^6 tons
19.9
Gas Losses
5.2
Infiltration
8.4
Leaching
RISING WATERS
Subsurface Flow
SUBSTRATA N POOL
40 x 10^6 tons
0.0
Recharge
Water Table
GROUNDWATER N POOL
Total = 58x10^3 tons
Satd. Zone = 49x10^3 "
Unsatd. Zone = 8.7x10^3 "
Pumped
5.0
Groundwater Flow

NITROGEN POOLS AND FLUXES

Figure 1. Nitrogen pools and fluxes in the 356,000-acre upper Santa Ana Basin based upon the 1960 level of development. The mass of nitrogen in the pools is in tons of nitrogen; numbers near the arrows represent the flux of nitrogen between pools in thousands of tons of nitrogen/year.

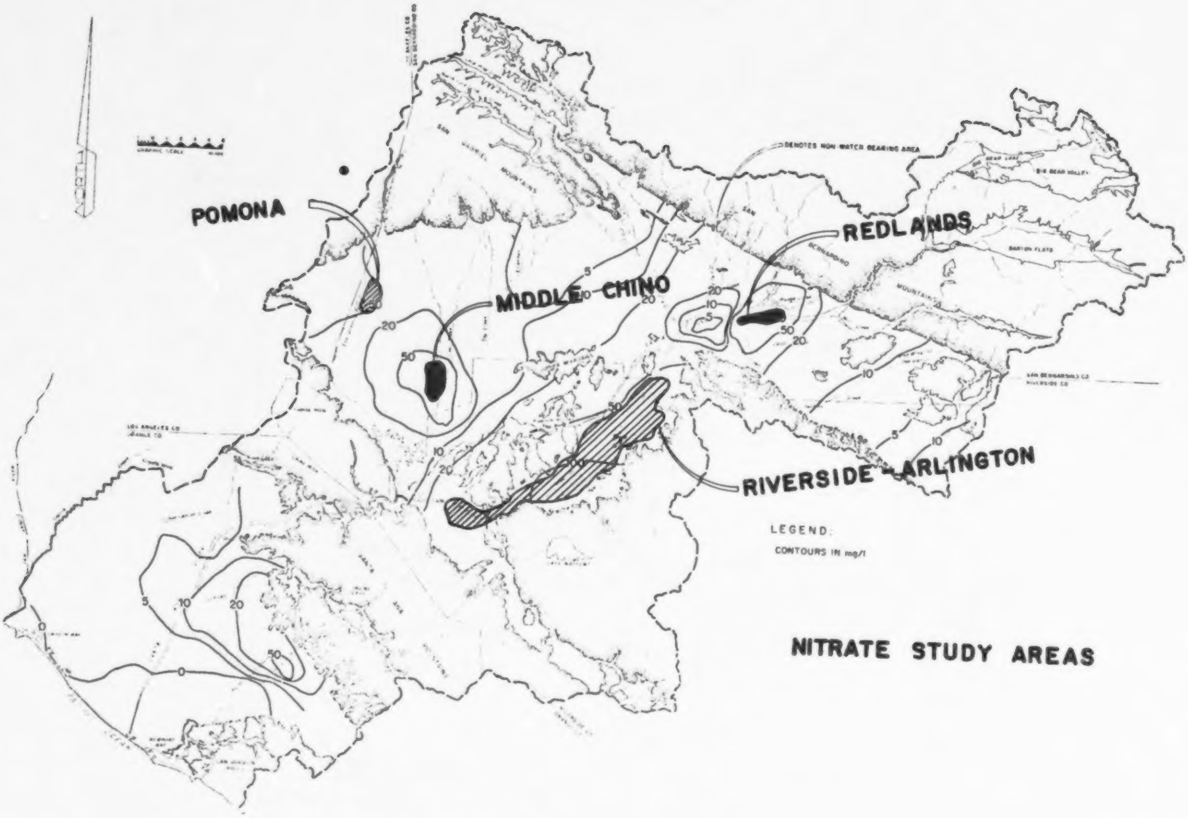

POMONA
MIDDLE CHINO
REDLANDS
RIVERSIDE - ARLINGTON
DENOTES NON WATER BEARING AREA
SAN GABRIEL MOUNTAINS
SAN BERNARDINO MOUNTAINS
BIG BEAR LAKE
BIG BEAR VALLEY
BARTON FLATS
SAN BERNARDINO CO
RIVERSIDE CO
LOS ANGELES CO
ORANGE CO
GRAPHIC SCALE
LEGEND:
CONTOURS IN mg/l
NITRATE STUDY AREAS
0
5
10
20
50

Figure 2. Map of the Santa Ana Basin showing locations of areas of high nitrate concentration. (Courtesy Water Resources Engineers, Inc.)

area of roughly 12,500 acres which included dairies and surrounding farm land available for disposal of dairy manure and barn washings. This equates to about 10 cows/acre.

4. Groundwater hydrology was studied in relation to nitrates.

   Data from existing reports showed that the groundwaters were, for the most part, unconfined and in hydraulic contact with the ground surface. Horizontal water velocities were slow in the areas of high nitrate (1 mile in 10 years to 1 mile in 300 years), and since the rate of vertical movement of nitrates through the nonsaturated soil before it reached the water table was also slow, this suggested that the high nitrates found may have reflected changes in surface nitrogen loading that occurred over a span of tens or perhaps hundred of years.

5. Three soil maps were prepared showing general soil associations and water transmission classes and their potential for denitrification.

   In general, nitrate concentrations were higher in groundwaters located below coarse textured good agricultural soils (loamy sands, sandy loams) under intensive agricultural use. But, not all such areas had high nitrates.

   It was suggested that soils of fine texture (clay loam and clay) may offer a useful potential for denitrification.

6. Water requirements and relative efficiency of use for several important crops were studied.

   For citrus, water use efficiency for surface irrigation varied from about 44% to 60% (LF = 48% to 25%) and efficiency did not appear to be appreciably better under sprinklers.

7. Nitrate concentrations in soil profiles were studied in relation to agricultural practices. This was new data.

   Fertilization of citrus at a rate of 150 pounds nitrogen/acre/year applied annually resulted in a nitrate nitrogen concentration of 19 mg/l in soil water below the crop; a rate of 350 pounds nitrogen/acre/year resulted in a concentration of 45 mg/l. Depth of sampling was 50 to 100 ft or to depth of water table. In general, the higher nitrate concentrations were associated with lower leaching fractions.

On vegetable crops, annual nitrogen rates varied from 100 to 1740 pounds nitrogen/acre. Nitrate concentration below the crop tended to increase as nitrogen rates increased, but factors such as variations in water application, denitrification, and crop removal as well as other factors often caused less nitrate to leach than expected. Because of these other factors nitrogen losses under vegetables were not as consistent or predictable as nitrogen losses under citrus. For example, for vegetables, the nitrate nitrogen concentration in soil water at the 11- to 50-foot depth ranged from 8 to 121 mg/l, whereas application rates varied from 100 to 1740 pounds/acre/year.

8. Nitrates in soil water and groundwater below the dairy farm area of the Chino Basin were measured by deep soil borings. Again, this was new data.

The amount of nitrate varied widely among sites. On the average, however, total nitrogen/acre for profiles through the 19-foot depth showed 1938 pounds of soluble nitrate nitrogen under the corral, 670 pounds under pasture, and 727 pounds under crop land fertilized with 20 to 45 tons of manure/acre/year. Nitrate nitrogen concentrations in the soil water of these same areas averaged 91, 74, and 65 mg/l. In contrast the water at top of the water table averaged 56, 74, and 45 mg/l and deep well waters in the same vicinity averaged 6 mg/l.

This data suggested that the dairy wastes had not yet had full impact on the quality of water in the saturated zone (water table).

A typical balance sheet for estimating the average nitrate nitrogen concentration of the water leaving the root zone under typical double-cropped manure disposal areas was prepared based on the number of cows/acre. In the calculations we assumed that 1) the nitrogen defecated was 0.4 pounds/cow/day, 2) 50% of the defecated nitrogen was lost to the atmosphere by volatilization of ammonia, 3) the nitrogen uptake by crops followed the usual pattern, and 4) that the drainage or percolating water was 15 inches/year (See Table 1).

Table 1 shows a nitrogen balance sheet for estimating the average nitrate-nitrogen concentration in soil water leaving the root zones under pastures and croplands. One-half of the nitrogen excreted is assumed to volatilize as ammonia, and the other half is incorporated into the soil. (Our analysis showed about 38% total nitrogen loss from corral manures, but more losses can be expected after land application). The excess in the soil is the amount incorporated into the soil minus removal by crops. This assumes that an amount of nitrogen equivalent to the amount of manure nitrogen incorporated into the soil each year was mineralized each year. This assumption is valid only after several decades of applications of manures and was probably justified only to obtain a rough idea of the amounts of nitrogen going into various sinks in relation to cow density. The rough estimates are the volume of drainage water, and the extent of denitrification and mineralization. The drainage volume used was 15 inches, which corresponds to an average leaching fraction (LF) of about 0.30, which is a common value for successful irrigation management. (Leaching fraction is the volume of water that moves past crop-root systems, expressed as a fraction of the total irrigation water used). The values for the 25% loss column were calculated on the assumption that one-fourth of the excess nitrogen is nitrified and then denitrified. Compared with some of the published data this assumption seems reasonable (Tisdale and Nelson, 1966).

Average nitrate-nitrogen concentrations in soil water at depths of 10 to 19 feet under the pasture and cropland sites were respectively 74 and 65 ppm. The concentrations, however, ranged from 24 to 196 ppm in pastures and from 15 to 210 ppm in croplands. Only 3 of the 18 sites

Table 1. Total nitrogen input in soils, removal by crops, excess in the soil, and calculated nitrate concentration in the soil water in the unsaturated zone, assuming two levels of loss of excess nitrogen.

Number of cows per disposal area	Total nitrogen excreted (all cows)+	Amount of nitrogen incorporated in soil	Amount of nitrogen removed by crops‡	Excess nitrogen in soil	Nitrate-nitrogen in unsaturated zone assuming*:	
					0% loss of excess nitrogen	25% loss of excess nitrogen
	----------------pounds/acre/year----------------				------------mg/1------------	
3.	438	219	190	29	8.4	6.3
4.	584	292	240	52	15.1	11.5
5.	730	365	270	95	27.8	21
6.	876	438	290	148	43	32
8.	1,168	584	320	264	77	58
10**. . . .	1,460	730	350	380	110	84
12.	1,752	876	370	506	148	111
14.	2,044	1,022	380	642	187	140

** The values for the nitrogen rate equivalent to the 10-cow level were based on 200 pounds nitrogen removal by 30 tons/acre of corn silage plus 150 pounds nitrogen removal by 4 to 5 tons/acre of green crops. In all cases, double cropping was assumed.

+ Assuming that each cow defecates 0.40 pounds nitrogen/day.

‡ Figures based on published data and field data of Extension Specialists in the area.

* In soil water in aerated zone below the root system, assuming the drainage volume to be 15 surface inches/year.

had nitrate-nitrogen concentrations of 20 mg/l or less in soil water at this depth. These variations were obviously due to differences in disposal rates, crop removal, leaching losses, and denitrification. Comparing these data with data in Table 1 we can see that some of the rates apparently exceeded the equivalent of 14 cows/acre.

Guidelines Suggested to Prevent Further Degradation

Rates of Fertilization. Because nitrogen is necessary for acceptable crop yields, the use of nitrogen fertilizers should be continued. For citrus the nitrogen should be applied in a single application using leaf analysis as a guide to the quantity to apply. Foliar feeding for this crop holds some promise for reduced nitrate leaching. For vegetable crops maximum yields require from 120 to 300 pounds of nitrogen/acre depending on the crop. Split nitrogen applications are recommended.

Use of Manures. Manures should be applied well ahead of planting and disced or otherwise incorporated into the soil. With small seeded crops an irrigation should be scheduled between incorporation of the manure and planting of the seed. For a number of crops a starter application of nitrogen fertilizer should be used. Manure should not be applied to growing crops. Washwater and liquid wastes should be applied at rates and at a time consistent with crop demands for water and nutrients.

Water Management. Excessive leaching losses can be reduced by reducing the amount of either the quantity of nitrogen applied or the volume of water applied. Overirrigation tends to reduce the concentration of nitrate in the drainage water but it decreases the efficiency of both nitrogen and water use. Nitrogen rates needed for adequate yields can be reduced (nitrogen use efficiency increased) by improving the efficiency of irrigation. Crop needs for water can be calculated or measured in various ways. Evapotranspiration can be used as a guide to irrigation, to estimate use efficiency and to estimate probable quantities of drainage water. Chloride balances can also be used to estimate the leaching fraction and the drainage volume.

Agricultural Waste Management. Wastes from agricultural operations (mainly dairies) should not be allowed to run-off into surface water. They should be used on cropland at rates consistent with recycling of nutrients and the excess should be exported from the basin.

At the time of the study waste disposal practices were not adequate. Specific proposals and suggestions were 1) the wastes from no more than three dairy cows should be used/acre to reduce nitrate leaching to a reasonable level, 2) restrict disposal of manure on lands of low underground pollution potential, 3) holding ponds to contain storm run-off should be sufficiently large to hold the run-off from a 10-year frequency storm, 4) each producer should be required to prepare a satisfactory plan for use or disposal of wastes, 5) zone the watershed for pollution potential based on soil characteristics, 6) relocate nitrogenous waste producers to areas with minimal underground pollution potential, 7) recharge underground with high quality water, 8) develop shallow wells for agriculture and deep wells for municipal supplies, 9) consider zoning a portion of the basin as a waste acceptor and recover water and nitrate for use in agriculture and 10) as a last resort, construct a "brine line" for collection, treatment and export of wastes from the basin.

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AN ECONOMIC METHODOLOGY FOR EVALUATING "BEST MANAGEMENT PRACTICES" IN THE SAN JOAQUIN VALLEY OF CALIFORNIA

Gerald L. Horner, Daniel J. Dudek, and Robert B. McKusick[1]

ABSTRACT

Section 208 of Public Law 92-500 (The Federal Water Pollution Control Act Amendments of 1972) requires the preparation of areawide waste treatment management plans. Agricultural related pollutants such as subsurface drainage water and irrigation tailwater containing nutrients, sediment and pesticides must be identified and procedures and methods to control such discharges specified. Under Section 208, the reduction of pollutants may be achieved by adopting a set of "best management practices" that could include varying economic incentives, establishing resource use controls and suggesting public resource investment. To determine the economic and environmental impact of "best management practices", a methodology must be derived that specifies the relationship of agricultural production practices to water and land quality and the economic cost and benefits of changing those practices. Such a methodology is being developed to systematically collect and organize physical and economic data, specify the physical and economic relationships and estimate the changes in agricultural production, income, employment and resource demands from proposed "best management practices".

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INTRODUCTION

The purpose of this paper is to: (1) establish the need for a comprehensive analytical system to evaluate water quality policies affecting irrigated agriculture; (2) delineate alternative "best management practices" (BMP)[2/] for irrigated agriculture; (3) discuss economic variables and relationships needed to evaluate Areawide Waste Treatment Management (208) plans; and (4) suggest a methodology and analytical system to evaluate BMPs and 208 plans for irrigated agriculture.

The Federal Water Pollution Control Act Amendments of 1972 (PL 92-500) and the Clean Water Act of 1977 suggest that water resource development, land use planning and environmental policies be coordinated, integrated and updated in a continuing planning process. This process is required by each state under two types of area specifications, designated and nondesignated. Designated areas are urban-industrial communities and the principle focus of the planning process is to coordinate non-point and point source waste disposal controls. Local planning agencies, and associated governments are responsible for developing designated area 208 plans.

Nondesignated areas require the identification of "agriculturally and silviculturally related non-point sources of pollution including runoff from manure disposal areas and from areas used for livestock and crop production" and the specification of "procedures and methods to control, to the extent feasible, such sources" (Section 208 (b) (2) (F)). The formulation of nondesignated area plans is the direct

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[2/] "Best Management Practices", are specified in PL 92-500, Section 208 as methods of achieving the comprehensive planning objectives in non-designated areas. BMPs are defined in the more limited sense to mean physical adjustments in the productive process or resource capacity to change the environmental condition.

responsibility of the state government. The Federal Government has allocated $600 million dollars in the next two years through the Culver Amendment to cost share applications of BMPs in those areas that will have the most significant effect on water quality.

Section 303 of PL 92-500 requires that the state adopt a continuing planning process that is consistent with all provisions of the Act. This process is designed to insure that the initial plan formulated under Section 208 remains effective under changing environmental conditions. In addition to its planning responsibility, each state must prepare an estimate of: (1) the environmental impact; (2) the economic and social costs necessary to achieve the objectives of this act; (3) the economic and social benefits; and (4) the date of achievement (Section 305 (b) (1) (D)).

Section 304(k) (1977 Clean Water Act) authorizes the Administrator of EPA to enter into agreements with other federal agencies to utilize their authorities and programs to support the development and implementation of state and local water quality management plans. This section recognizes that implementation of many state and local programs is dependent upon the expertise and support of other federal agencies. Traditionally, most agricultural programs have been conservation or production oriented. The extent that past conservation and production practices will become "best management practices" is a subject of current debate.

At the present time, a clear set of alternative "best management practices", implementation procedures and evaluation criteria do not exist. To systematically evaluate 208 plans and BMPs, definite alternative strategies and evaluation criteria must be specified prior to the formulation of an analytical system. The list of alternative BMPs and economic parameters specified in this paper is not intended to be complete but it does include some practices and economic implications not currently being considered by 208 planners.

The BMPs implementation incentives and the institutional arrangement for 208 plan implementation are interdependent. The selection of a particular control strategy must be a combination of practices and incentives tailored to the geographical setting, the pollution problem and the responsible institution (Bower et al, 1977).

ALTERNATIVE BEST MANAGEMENT PRACTICES

Best Management Practices can be specified for agriculture, silviculture, industrial or municipal operations to decrease the impact of non-point sources of pollution on land and water resources. The BMPs discussed here are limited to irrigated agriculture and the impact of irrigation return flows on agricultural land and ground and surface water supplies. The BMPs for irrigated agriculture can be broadly classified into four categories: (1) reducing the quantity or improving the quality of irrigation return flows; (2) treating or reusing irrigation return flows; (3) improving irrigation return flow disposal practices to minimize the environmental impact; and (4) increasing the capacity of the receiving resources to assimilate the irrigation return flows.

REDUCING IRRIGATION RETURN FLOWS

Reduction of the quantity or improving the quality of irrigation return flows by modifying cultural practices has received most of the attention of point and non-point pollution control institutions. One control practice is to change the amount of inputs used in the agricultural production process. Reducing the amount and increasing the quality of irrigation water, fertilizer and chemicals are prime examples of procedures to reduce the toxic chemicals, nutrient content, sediment content, salinity content, and volume of irrigation return flows. These general relationships are fairly well established as pollution control practices (USEPA, 1973 and USDA-USEPA, 1975).

Another method of improving return flow quality is to change the method and timing of water, fertilizer and chemical applications to either maintain the same production with less inputs or improve production with the same amount of inputs. Both procedures will generally reduce the environmental impact of irrigation return flows.

Changes in the cropping pattern or type of agricultural use can drastically affect the amount of polluted water entering a watercourse. In areas possessing high soil erosion potentials, row crops and orchards could be replaced with crops that provide better surface protection. Replacing row crops with irrigated or non-irrigated pasture would be an extreme but perhaps necessary land use change to reduce erosion to an acceptable level.

TREATING OR REUSING IRRIGATION RETURN FLOWS

Once irrigation return flows are produced, they can be reused on the field where they were generated, reused on the farm, reused within the irrigation district or region, treated to remove contaminants or disposed of in the watercourse. The reuse of surface irrigation return flows is currently practiced in most areas in the west where it is convenient to do so. Many economic and institutional questions regarding the rights of irrigators currently relying on irrigation return flows will need resolution before extensive recycle programs can evolve as a workable solution. Reusing irrigation return flows to cool electric generation plants and/or to maintain wildlife habitats is currently occurring in areas of the Central Valley of California. The possibility of expanding these uses exists.

The amount of irrigated land requiring subsurface drainage is projected to increase over the next fifty years (U.S. Department of Interior, 1969). The quality of irrigation return flows over time will depend on the proportion of subsurface to surface drainage water.

In the early 1970's, the California Department of Water Resources, the U.S. Bureau of Reclamation and the Environmental Protection Agency explored the physical and economic possibility of removing nitrogen and salts from subsurface irrigation return flows. Results from that study indicate that removing most of the nitrogen from return flows cost about $44.00 per acre foot and removing salts would cost about $100.00 per acre foot (1971 Dollars) (California Department of Water Resources, 1971).

IMPROVING RETURN FLOW DISPOSAL PRACTICES

Watercourses receiving irrigation return flows have a natural capacity to assimilate nutrients, chemicals and sediment without impairing subsequent use or enjoyment of the water. However, when the natural assimilative capacity is exceeded, conflicts among the firms or persons affected by the decrease in quality usually occur. Changing the spatial or temporal distribution of return flow disposals will take advantage of the assimilative capacity of the environment without affecting agricultural production. This change in distribution can be accomplished in areas such as the San Joaquin Valley by collecting return flows with high pollution concentrations and storing them for release in high natural flow periods or transporting them to areas that have sufficient assimilative capacity.

INCREASING THE ASSIMILATIVE CAPACITY OF THE RECEIVING WATERCOURSE

A good example of increasing the assimilative capacity of the environment was demonstrated last year when releases from the State Water Project were made to decrease the salinity concentration of the Delta. Releases of stored water in normal rainfall years to achieve minimum water quality standards is one alternative to accomodate increased irrigation return flows.

INCENTIVES FOR IMPLEMENTING BEST MANAGEMENT PRACTICES

The states face a formidable problem in implementing control strategies for non-point pollution. Most methods to control point sources consist of "end-of-pipe" treatments. Most non-point pollution problems are not amenable to treatment and those that are cannot be associated with any one emitter. Consequently, responsibility for the cost of treatment is difficult to assign. There are many alternative implementation incentives and they vary in effectiveness, costs and institutional and political requirements. Incentives include both positive and negative values to producers and resource owners. The proper set of incentives and control methods is dependent on the type of agriculture and pollutant. Five classes of implementation incentives relating to irrigated agriculture are delineated by Bower (1977). They are regulatory, economic, administrative, judicial and educational.

INCENTIVE BY REGULATION

Many point source pollution programs employ regulation to control one or more aspects of the productive or waste emission process. Regulations are usually effective in achieving point source pollution standards but the same success may not be realized in non-point control. The relationship between changes in the productive process and water quality is usually lagged by a substantial period of time and is difficult to measure because of numerous uncontrolled factors.

The specification of cultural practices in irrigated agriculture to reduce return flows would consist of limiting the amount or type of productive inputs, limiting the timing and method of polluting activities and limiting crops that generate unusually large amounts of pollutants. Other specifications might include land use controls, collection and separation of subsurface drainage water, implementation of structures to contain or reduce the pollution potential of return flows and limiting the quality of irrigation return flows for disposal. These specifications are well known to farmers and environmental planners. However, some of the institutional and economic considerations of regulation may not be so well known. These will be elaborated in the discussion on evaluation criteria.

ECONOMIC INCENTIVES

In contrast to regulating agricultural activities, financial aids and penalties which apply to emission of pollutants can potentially achieve an environmental standard at a lower cost than regulations because the farmer has a choice of actions. A charge on pollutants discharged to a watercourse allows the discharger to reduce emissions if the cost of doing so is less than the charge. However, the effluent charge system is difficult to apply to non-point polluters as emissions from each farmer have to be measured. Even in point source control programs, charges on pollution emissions have never received the political support required for implementation in the United States.

Financial incentives can also be placed on the use of specific inputs. Reducing the nitrogen content of irrigation return flows may be accomplished by taxing fertilizer. Farmers, if faced with a substantial increase in fertilizer costs, could employ more efficient application and timing techniques to reduce fertilizer use and maintain production.

Increasing the efficiency of irrigation water conveyances, delivery and application systems will generally reduce irrigation return flows and mass emissions of nutrients, sediments and salts. High water use efficiency is usually observed where water prices are high or water allocations closely approximate crop requirements.

In recognition of this fact, the Governor has established a commission to review the Water Rights Laws of California. The purpose of the commission is to suggest any needed legislation which would improve the water allocation system. Lee (1977) has explored the possibility of allowing water rights to be transferred by a competitive market. The effect of such a scheme would be a general increase in the value of water in those areas that are currently experiencing low prices. Gardner and Fullerton (1968) concluded that allowing water transfers reduced the risk of insufficient supplies, increased effective water supplies and resulted in a more efficient use of water resources.

Recent research suggests that increasing the price of water will have a dramatic impact on reducing the quantity and improving the quality of irrigation return flows. In a study conducted in the Yakima Basin of Washington, Pfieffer (1976) concluded that increasing the price of irrigation water was the least costly method of achieving temperature, sediment and nutrient standards on the Yakima River. Horner and English (1976) concluded that increasing the price of surface irrigation water would reduce the total salt loading and quantity of return flows in the San Luis Drain.

A bill recently introduced in Congress reflects this same theme (Miller et al, 1977). The bill proposes that a base volume of water be established for areas using Federal water based on crop requirements. Additional water supplies in excess of the base volume would be available at increased block rates.

Land use directly affects the demand for water resources and the quality of water within a region. One example of using economic incentives to achieve a desired land use is a preferential property tax system. The California Land Conservation Act (Williamson Act) employs a use-value assessment mechanism to base taxes upon income rather than market value. The nature of non-point pollution from an agricultural area differs from that from the urban setting and different controls need to be considered in the planning process.

PROVIDING INCENTIVES THROUGH EDUCATION

Programs to extend knowledge about reducing pollution can be an effective method of reducing irrigation return flows if the proposed actions are in the best interest of the irrigator. Convincing a person that his actions will benefit society and yet cost him money, is usually a difficult task for the best extension person. However, a substantial portion of the pollutants currently found in irrigation return flows result from irrigators not managing their systems as efficiently as possible. These systems could be improved at low costs.

JUDICIAL INCENTIVES

Many of the present public policies and the administration of these policies regulating the environment are a result of previous judicial decisions. Recently a court decided that an environmental impact statement will be required on the proposed rules to enforce the 1902 Reclamation Act. In other cases, "cease and desist" orders against polluters have been granted to injured parties. Although the courts do provide an alternative implementation incentive, many disadvantages exist. First, courts are usually slow in rendering decisions and given the appeals process, damages from emissions may not be stopped soon enough. Second, the persons being damaged by pollution are not usually sufficiently organized to bring court action.

CRITERIA FOR EVALUATING BEST MANAGEMENT PRACTICES

The approval of the Section 208 plan and the related BMPs will depend on several criteria. The approval mechanism provides organized groups and individuals with the opportunity to participate in the final review of the plan. That review process is expected to require diverse sets of information. The following criteria were suggested by Bower et al (1977) specifically for evaluating alternative environmental strategies.

ENVIRONMENTAL OR PHYSICAL EFFECTS

The degree to which an environmental strategy can improve the overall quality of resources is the ultimate criteria. The quality of resource will affect other physical conditions such as improved wildlife habitats, outdoor recreation opportunities or decreased water treatment costs. Irrigation return flows either terminate in surface watercourses or underground aquifers. Measuring the quality and quantity of return flows prior to being incorporated with the receiving waters would be an appropriate measure of control method effectiveness but it would not determine the degree of improvements in the quality of the watercourse. The relationship between changes in irrigated agricultural practices and the quantity and quality of irrigation return flows is not completely known for a very broad set of physical conditions. Therefore, a certain amount of guesswork will be involved in determining appropriate BMPs. Even less is known about the relationship between quantities and qualities or return flows, overall water quality, and damages to fisheries, wildlife or persons using the watercourse. These relationships need to be measured before control strategies can be correctly evaluated.

ECONOMIC EFFECTS

Direct Benefits and Costs

Direct benefits from improved water quality can be measured in reduced water treatment costs, increased value of a fishery, reduced medical costs, and reduced damages to property. Direct costs are those incurred by farmers as a result of the control method. These include the costs of capital, labor, land, management, water and reduced production should it result from the control method. Other direct costs include the cost to the public agency of providing facilities to reduce return flows or to improve the assimilative capacity of the environment.

Indirect Benefits and Costs

The modern industrial economy is highly interdependent. Reductions or expansions in a given industry will result in changes in the amount of resources purchased from other industries and supplied to other industries. Depending on the degree of interdependence, the indirect effects can exceed the direct cost and benefit to a single industry in terms of income, employment and resource use.

Administrative Costs

Implementing control strategies usually requires additional accounting, monitoring, reporting, supervision, enforcement and management. These additional costs vary substantially according to the type of control method selected and they are usually borne by public agencies and irrigators alike. The importance of administrative costs is usually not considered in deciding on a course of public action. In some cases, the public and private administrative costs of a program could exceed the public benefits derived from the program (Seckler, 1965).

The Distribution of Benefits and Costs

Persons or firms benefiting or bearing the costs of a pollution program should be identified by geographical area or socioeconomic group. Horner and Dudek (1977) concluded that increasing the cost of irrigation water to reduce irrigation return flows reduced the income to small farmers proportionately more than large farmers. Bower et al (1977) characterize the distribution of benefits and costs as the most important physical and economic effect of pollution control. However, measurement of the distributional effects of public policies is difficult due to the confidential nature of personal income and asset data.

Ease of Administration

An effective control strategy must be flexible enough to accomodate changing economic conditions. Included in this criteria is the ability of the method to self adjust to changing conditions. The ability of one control method to be applied to various pollution parameters and activities is a desirable characteristic. The procedural ease of adjustment and application is also important. For example, a permit system applied to irrigation return flows requires considerable effort and time to file applications and establish monitoring positions.

Political Considerations

The 208 plans require approval from a diverse set of interest groups. Each interest group representative will probably weigh the effects of each BMP on the goals of the interest group. This approval process suggests that compromise BMPs will probably result. The politics of governments may also be important. The responsibility for the Sacramento-San Joaquin River Delta water quality is a case in point. Although the Bureau of Reclamation and the State Water Project use the Delta to transport water to the San Joaquin Valley and Southern California, the Bureau does not accept responsibility for maintaining the level of quality as the State does. In addition, the institutional arrangement and legal framework may not exist to allow some control strategies to operate. The establishment of new institutional structures represents an important political decision requiring general support of the public. The last, but certainly not the least, political consideration is the acceptance of the program by the general public.

Interdependence of Land and Water Quality

The National Water Commission's call for closer ties between water resources planning and land use planning and their subsequent integration with water quality policy is explicitly recognized by PL 92-500. Currently, opportunities exist for the integration and simultaneous completion of several water resource planning activities. Water and related land resource planning is required under the Water Resources Planning Act (PL 89-80), the Watershed and Flood Prevention Act (PL 83-566), the Colorado River Basin Salinity Control Act (PL 93-320), and the Federal Water Pollution Control Act Amendments of 1972 (PL 92-500). In particular, PL 92-500 requires planning by states (Sections 303 and 305), river basins (Section 209), and areas within states (Section 208). These individual planning requirements are not currently integrated or coordinated. Section 208 of PL 92-500 provides an opportunity to integrate water quality planning with water resource development and land use planning.

Current and future conflicts in resource policies must be identified. One potential policy conflict is between the development of agricultural preserves of prime agricultural land and non-point source controls. Agricultural preserves could concentrate production and thus compound non-point source pollution output. However, the environmental impact of the point source discharges from urban development may be greater. There could also be a conflict between maintaining agricultural lands for food production and improving the economic viability of agricultural sector (through tax advantages), open space, and non-point source pollution. These issues would involve a conflict between national and regional economic objectives and environmental quality objectives.

Other questions relate to the relationship between productive capacity and projected demands facing the agricultural sector. One report concludes that the "consideration of adverse environmental impacts from agricultural production activities may ultimately prove to be the major constraint on resource development in the next 24 years" (USDA-Economic Research Service, 1976). Crosson (1975) amplifies these concerns in a consideration of the kinds of technology that may be used to generate increased agricultural production. He classifies the technological alternatives as either land-using or land-conserving. Land-using technologies result in loss of wildlife habitat and soil loss due to erosion with resulting increases in turbidity and phosphate fertilizer transport. The environmental damages associated with land-conserving technologies are increased loadings of fertilizers and pesticides.

THE METHODOLOGY OF EVALUATING BEST MANAGEMENT PRACTICES

Resource and environmental planning cannot be separated. Planning for resource use without recognition of the environmental goals or objectives of society may result in resource allocations which are socially suboptimal. Similarly, planning for environmental quality without assessing the suitability, availability and productivity of the resource base may impair economic efficiency and distribution of output.

The economic information required for resource planning and evaluation can be discussed under four interrelated headings; commodity demand, commodity supply, resource demand and resource supply. Given the planning setting, the necessary set of information and the analytical models to be employed, it is useful to see how the basic economic concepts of commodity demand and supply and resource demand and supply will be operationalized. These concepts can readily be identified as flows of information between different components of the analytical system as indicated in Figure 1. This analytical system is being developed for the San Joaquin Valley research project funded by the Environmental Protection Agency and the Economics, Statistics, and Cooperatives Service, USDA.

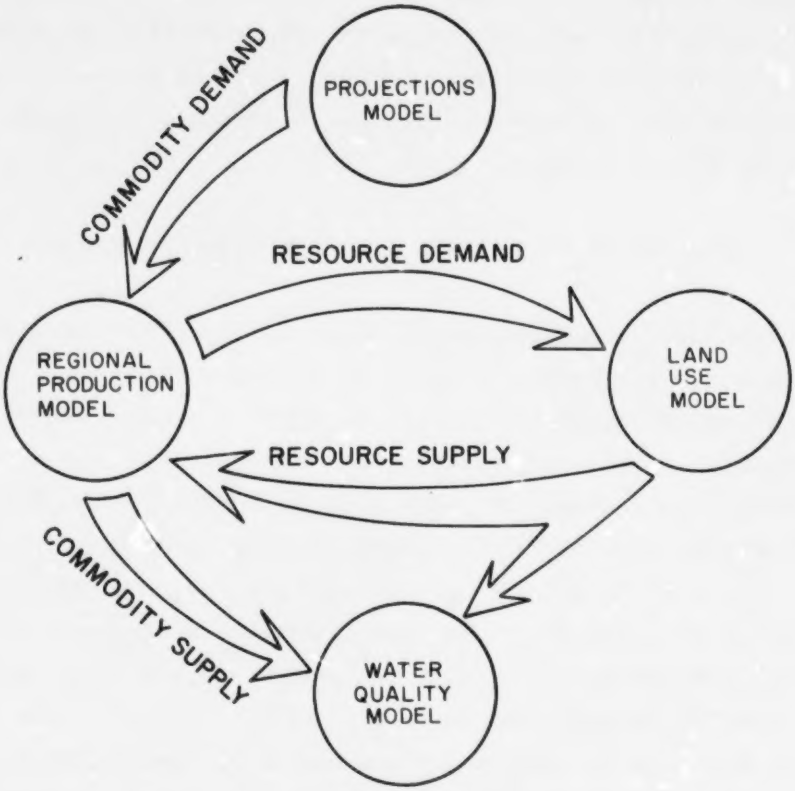
COMMODITY DEMAND
PROJECTIONS MODEL
RESOURCE DEMAND
REGIONAL PRODUCTION MODEL
LAND USE MODEL
RESOURCE SUPPLY
COMMODITY SUPPLY
WATER QUALITY MODEL

Figure 1. Interactions among components of the irrigated agricultural analytical system.

The demand for regional commodity production is projected from estimates of changes in population, income and U.S. trade policies. The production model determines the amount of land and productive inputs to achieve the projected level of commodity demand which in turn is effected by land use policy decisions. The specific location of commodity production and resource use is determined by the water quality model and projections of the amount and quality of irrigation return flows result. Alternative BMPs can be simulated in the appropriate model to determine the effect on the criteria just discussed. Each model will be explained below in more detail.

COMMODITY DEMAND

The process begins with the specification of commodity demand from the projections model. In order to accurately portray changes in the irrigated crop economy of the valley between periods, projections of changes in technological coefficients, yields and the demands for crop commodities facing the region are needed. Particularly crucial to the determination of resource problem effects is the projection of commodity demands. While there is a wide variety of projection approaches varying with the quantitative techniques employed, the area of interest and their ultimate application, the basic approach employed is that reported by King et al (1977).

Essentially, projections of national demand are disaggregated to the state or regional level through the projection of share of national production as in Dean (1970). The principal shifters of the domestic component of national demand are population and per capita income (Dean and King, 1970), but trends in per capita consumption, presumed to capture changes in tastes and preferences, may also be included. The addition of projected export-import balances produces estimates of national demand requirements, i.e., given favorable economic conditions, no radical changes in price-cost relationships and the assumption of perfectly elastic supply, these are the commodity quantities which would have to be produced in order to satisfy the domestic and export demands. Thus, these projections are conditioned upon the levels specified for key parameters -- population, per capita consumption, per capita income, exports and imports.

COMMODITY SUPPLY

Commodity supply response is estimated by the Regional Production Model using the San Joaquin Valley Basin regional linear programming model. This model has been developed by the USDA California River Basin Planning Staff as part of an ongoing analysis of the water and related land resource problems of the area (McKusick, 1973 and McKusick, 1974). The linear programming model was developed to analyze the impact of deteriorating drainage conditions on production patterns in the study area and the evaluation of program, policy or project measures designed to remedy the problem. The model is static, i.e., designed to produce solutions for a single specified period. The objective function maximizes net agricultural crop returns in the basin subject to the availability of land resources by soil group, drainage condition and location, the availability of surface and groundwater for irrigation, labor, crop production (or crop acreage) restrictions and the underlying production technology.

The San Joaquin Valley is divided into two component subbasins, each separately modeled. Each subbasin is subdivided into an east and west side on the basis of differing salinity and drainage characteristics. For 30 principal crops, production activities differ by soil group (of which there are 18), drainage condition (three discrete categories) and location. Technological coefficients on an annual per acre basis are specified for yields, applied water requirements (which include estimates of the leaching requirement and embody an implied irrigation efficiency), harvest and nonharvest labor, fertilizer (in nitrogen equivalent terms) and gas and diesel fuels.

RESOURCE DEMAND

The regional linear program will be used to generate the demands for those land and water resources. These resource demands are the basis of land use valuationswithin the control model. The optimal control model will be used to simulate the dynamics of changes in land uses over time for the valley in reaction to changes in agricultural commodity demands, population, resource productivity and availability and environmental and land use policies. The control model will generate optimal land uses over time, the social values resulting from these allocations and the opportunity costs of changing those resource uses.

RESOURCE SUPPLY

The control model, in turn, predicts the availability of land resources for agricultural production which are required in the regional linear program. Projections of commodity demands from the system previously described are introduced into the linear program and a new cycle of model interactions is initiated. In this manner, the process is repeated over the planning horizon with the result being a close approximation of the optimal patterns of land uses over time in the Valley.

Resource and commodity supplies are also utilized by the water quality model. This analytical subsystem consists of two specific models sequentially linked to simulate agricultural production and environmental adjustments that occur as a result of an environmental policy (Horner and English, 1976). The first is a linear programming model that derives optimal cropping pattern, water application technologies, water use and fertilizer in 300 subregions of the valley. The production patterns derived in the linear programming model serve as data inputs to the physical model. The physical model estimates the subsurface vertical and horizontal water movements and surface runoff created by the irrigation activity. The costs for collection and disposal of return flows, the costs for installing tile drainage to relieve high water tables, and yield reductions from high water tables are also calculated by the physical model. The change in production costs for subsequent periods are then adjusted in the linear programming model. Solutions from the models are derived annually and are iterated a sufficient number of times to simulate the environmental adjustments from a change in water and land use as a result of alternative control policies.

INFORMATION AND DATA REQUIREMENTS

The analytical system requires information about the economic and physical systems of the region. Table 1 relates specific data requirements with each analytical system. The physical data is used extensively by the water quality model and to a lesser degree by the land use and regional production models. The bulk of the data has been collected from the U.S. Geological Survey, California Department of Water Resources, U.S. Bureau of Reclamation, U.S. Soil Conservation Service, University of California and other published sources. The data have been computerized to accomodate the common data requirements of these models.

The economic data are used principally to project the regional demand for agricultural production. Resource and commodity price data are used by all models. Most of the economic data was collected from publications of USDA, Bureau of Census, California Department of Food and Agriculture and California Counties located in the study area.

Table 1. Data requirements for the analytical systems.

Data Category	Projections Model	Water Quality Model	Land Use Model	Regional Production Model
Physical Data				
Unconfined Aquifer				
Depth to		x	x	
Quality		x	x	
Amount Pumped		x		x
Confined Aquifer				
Depth to		x	x	
Quality		x	x	
Amount Pumped		x		x
Surface Water				
Flow or Runoff			x	
Quality			x	
Amount Diverted		x	x	x
Precipitation		x	x	
Soil Characteristics				
Acreage		x	x	x
Location		x	x	x
Productivity		x	x	x
Resource Use by Crop		x		x
Porosity		x		
Permeability		x		
Specific Retention		x		
Geological Formations		x	x	
Land Use by Location		x	x	x
Economic Data				
Commodity				
United States				
Acreage	x			
Production	x			
Exports	x			
Imports	x			
California				
Acreage	x			
Production	x			

Table 1. Continued

County				
Acreage	x			x
Production	x			x
Prices	x		x	x
Input Prices	x	x		x
Population	x		x	
Tax Rates			x	
Ag Land Preservation Programs			x	
Drainage Installations			x	

SUMMARY AND CONCLUSIONS

Why should a regional comprehensive analytical system be developed to evaluate water quality policies affecting irrigated agriculture, in particular 208 plans and BMPs? As Bower et al (1977) point out, the degree to which an environmental strategy can improve the overall environmental quality is the ultimate policy criteria. The physical, environmental and economic impacts of implementing BMPs and 208 plans are not known but society has to anticipate these impacts in order to develop workable national and local water quality policy.

By 1979, 208 plans are to be developed and implementation started. By 1983 the nations waters are to be fishable and swimmable and by 1985 there is to be zero discharge of pollutants to these waters. The potential physical, environmental, and economic impacts (both on-site and off-site) of implementing BMPs and 208 plans have generally been ignored by resource planners. Information on these impacts is needed to determine if BMPs will be adopted in irrigable areas, the impact on the quality and quantity of irrigation return flows and the feasibility of the 1979, 1983, and 1985 goals.

The paper did not present empirical results. An anlytical system to quantify impacts of BMPs for irrigated agriculture, on resource use and irrigation return flow has been presented. It recognizes that successful implementation of BMPs and 208 plans cannot exclude other conservation and production practices. The scope of analysis involves not only the structure and operation of the individual farmer but the agricultural economy of the watershed and/or river basin where BMPs are implemented. Economic impacts of proposed environmental strategies should recognize the direct and indirect benefits and costs of improved water quality, program administration and other institutional costs, the distribution of costs and benefits, and the implications of their distribution for cost-sharing and penalties.

Water quality policy should not be developed in isolation of land, air, conservation, production and other water policies. A model or mechanism to analyze all of these policies simultaneously does not exist and it is questionable if such a system will ever exist. The comprehensive irrigated agriculture analytical system presented in this paper is a step in the right direction because we allow the interaction of national commodity demands with regional production, land use and water quality. The model does require a significant amount of data, both in quantity and detail. In the future the sensitivity of various environmental strategies to model specification and data aggregation will be tested. The sensitivity analysis will allow for a more generalizable model and data set that could be applied to other areas of the West.

In the future, it might be possible to address such irrigated agriculture policy issues as: the trade offs between BMPs, and conservation and production practices; the conjunctive management of surface, ground, return flow and recycled water; the economic definition of prime agricultural lands; the productive supply potential of the San Joaquin Valley given resource suitability, availability and quality; and the evaluation of proposed changes in water rights.

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NITROGEN BALANCES FOR THE SANTA MARIA VALLEY

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ABSTRACT

A simplified steady-state model has been used to develop nitrogen balances in the Santa Maria Valley, California. Balances have been developed for selected management units and for the valley as a whole. The balance developed for a field cropped with vegetables for the past 12 years showed that 30% of the applied nitrogen was removed in harvested crop, 37% was leached below the root zone and 33% was unaccounted-for which was attributed to gaseous losses of nitrogen as products of denitrification. Direct field measurement of denitrification over an eight-month period at one site in the same field found a 29% loss of the applied nitrogen. The first estimation of the nitrogen balance in the valley attributed 24% of the applied nitrogen to removal in harvested crops, 39% to leaching and 37% to denitrification.

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INTRODUCTION

A decision was made in 1975 to include a basin study in the last two years of a project entitled "Nitrates in effluents from irrigated lands" financed in part by NSF. This basin would serve as a focal point for findings from earlier research on this project as well as contribute to a better understanding of nitrogen sources and sinks within agricultural production systems. After discussions by project personnel, the Santa Maria Valley was selected for study. Some of the reasons for selecting this valley were that it is a well defined land area, a variety of field and vegetable crops are grown there, some information about nitrates leaching in the valley was available, and cooperation of farmers and extension service personnel was assured.

The Santa Maria Valley is a coastal valley located along the Santa Maria River at the boundary of Santa Barbara and San Luis Obispo Counties in California. The Santa Maria Drainage Basin encompasses approximately 67,000 ha (Hughes, 1975) of which 18,600 ha are intensively farmed with field and vegetable crops. The flat bottomlands and adjacent terraces are used for irrigated crops whereas the surrounding foothills are used for rangelands and some dryland farming.

One of the objectives of the "Santa Maria Project" was to develop data for nitrogen balances for the valley. These balances were to be on two levels, one for the valley as a whole and the other for a number of individual management units (fields) within the valley. The techniques used to accomplish this objective will be discussed in this paper along with data from one management unit to serve as an example of results from the project (eight units were included in the study). The nitrogen balance for the entire valley is still not complete and thus only the first approximation of the balance will be discussed.

The second objective of the project was to develop techniques to determine denitrification directly in the field. Results of this research are presented elsewhere (Ryden et al, 1978a, 1978b), however, some reference will be made to results obtained for one management unit.

APPROACH

A steady-state conceptual model that has been described by numerous authors, including Fried et al (1976), served as the basis for the studies carried out in the Santa Maria Valley. Basically this is an input-output model with the underlying assumption that at steady state the net change in the organic nitrogen pool over the period of study is zero. The greater the length of a study and the more uniform the management variables over time for an area, the greater the probability that this assumption will be fulfilled. For the management unit considered here, 12 years of cropping history were available. With certain assumptions, this information was integrated with the nitrogen input and output data collected over the two-year study period to develop a nitrogen balance for the 12 years.

The nitrogen inputs considered in this model were nitrogen added as fertilizer and in irrigation waters. The sinks for nitrogen considered in the model were removal in harvested crops, leaching and denitrification. Other inputs (e.g. nitrogen in rainfall, dust, etc.) and outputs (e.g. erosion, runoff, etc.) were considered to contribute little to the overall balance and the assumption was made that they cancelled each other out.

For individual management units, the information on fertilizer nitrogen inputs was obtained from records of management history that were supplied by farmer-cooperators. Nitrogen inputs in irrigation waters were based on measured nitrate-nitrogen concentrations in the waters and records of water application in the various fields.

During 1976-1977, the yields and nitrogen contents were determined for a variety of crops in the valley to serve as a basis for determining crop removals of nitrogen (Lund and Pratt, 1977). Average nitrate-nitrogen concentrations in soil solutions leaching below root zones of typical fields in Santa Maria were determined by soil sampling. These were used in conjunction with leaching volumes as obtained from a chloride balance model to determine nitrogen leached (Pratt et al, 1978). The soil sampling program was also designed to examine the spatial and temporal variation in nitrate-nitrogen leaching for the selected management units (Lund et al, 1978). One estimation of the denitrification-volatilization sink was obtained by balancing inputs with crop removals and leaching with the "unaccounted-for" nitrogen attributed to denitrification-volatilization. Denitrification was also measured in the field as reported by Ryden et al (1978b) and the data compared to those obtained by the balance approach.

BALANCE FOR SIMAS FIELD

As an example of a nitrogen balance for a defined management unit, data obtained for the Simas study area will be discussed in this section. This field was selected for study because it was typical of the heavier-textured, high organic matter soils in the western part of the Santa Maria Valley where vegetable crops dominate the irrigated agriculture. The Simas field had been planted to vegetable crops for at least the past 12 years. Data for the Simas field relative to crop type, fertilizer and irrigation water applied and yield as supplied by the farmer-cooperator are given in Table 1. The field was divided into three blocks, each 4 to 5 ha in size. A number of important points can be made relative to the data in Table 1. Typical of vegetable crop production in California, large amounts of water and fertilizer nitrogen were applied. Also more than one crop was typically grown each year on lands used for

Table 1. Cropping history of Simas field. Yields reported here are based on material leaving the field. For lettuce, field weight is equivalent to market yield; whereas, for cauliflower marketable yield is approximately 34% of field weight, and for broccoli and clery it is 80% of field weight.

		Block A		
Year	Crop	Water	Fertilizer nitrogen	Yield
		ha-m	kg/ha	T/ha
1976	Broccoli	.91	232	14.3
	Lettuce	.73	125	51.7
1975	Broccoli	.85	242	15.7
	Broccoli	.82	197	17.9
1974	Cauliflower	.91	416	40.2
	Lettuce	.67	218	44.7
1973	Seeded celery	1.10	469	84.2
1972	Broccoli	.67	239	14.7
	Seeded celery	1.10	348	92.2
1971	Broccoli	.85	194	14.2
	Seeded celery	1.07	432	92.4
	Broccoli	.76	171	14.7
1970	Lettuce	.73	370	35.6
1969	Celery	.79	671	93.0
1968	Cauliflower	.91	298	12.4
	Celery	.82	651	69.4
1967	Broccoli	.76	153	18.6
	Celery	.76	608	68.6
1966	Broccoli	.70	176	17.2
	Lettuce	.73	183	48.1
1965	Celery	.79	659	100.1

Table 1. Continued

Block B				
Year	Crop	Water	Fertilizer nitrogen	Yield
		ha-m	kg/ha	T/ha
1976	Broccoli	.91	232	14.3
	Lettuce	.73	116	62.8
1975	Broccoli	.85	236	15.7
	Seeded celery	1.04	283	72.3
1974	Cauliflower	.91	290	41.2
	Lettuce	.67	220	45.7
1973	Seeded celery	1.10	393	105.7
1972	Broccoli	.67	176	18.6
	Lettuce	.73	175	39.2
	Broccoli	.76	227	12.7
1971	Seeded celery	1.07	351	81.6
1970	Broccoli	.73	227	15.8
	Lettuce	.73	357	35.6
1969	Celery	.79	448	93.0
1968	Cauliflower	.91	310	12.4
	Celery	.82	596	69.4
1967	Broccoli	.76	153	18.6
	Celery	.76	557	68.6
1966	Broccoli	.70	194	17.2
	Lettuce	.73	200	53.9
1965	Celery	.79	627	87.5

Table 1. Continued

Block C				
Year	Crop	Water	Fertilizer nitrogen	Yield
		ha-m	kg/ha	T/ha
1976	Broccoli	.91	232	14.3
	Lettuce	.73	127	44.1
1975	Broccoli	.85	236	15.7
	Seeded celery	1.04	289	72.3
1974	Cauliflower	.91	290	41.2
	Lettuce	.67	243	42.5
1973	Seeded celery	1.10	399	84.2
1972	Broccoli	.67	176	18.7
	Lettuce	.73	164	38.9
	Broccoli	.76	227	12.7
1971	Seeded celery	1.07	530	64.3
1970	Broccoli	.73	171	14.7
	Lettuce	.73	307	35.6
1969	Celery	.79	562	80.6
1968	Cauliflower	.91	299	12.4
	Celery	.82	630	69.4
1967	Broccoli	.76	148	18.6
	Celery	.76	485	68.6
1966	Broccoli	.70	207	17.2
	Lettuce	.73	160	49.6
1965	Celery	.79	577	83.3

vegetable crops. For this field 21 crops were grown in 12 years. These data show that over the past 12 years, 7053 kg fertilizer nitrogen/ha had been applied to Block A, 6368 kg to Block B, and 6459 kg to Block C.

Amounts of water applied and nitrogen concentration of that water were needed to calculate the amounts of nitrogen applied in the irrigation water. For the Simas field, 17.43 ha-m/ha of irrigation water had been applied to Block A over the past 12 years with 17.16 ha-m/ha being applied to Blocks B and C. All three blocks were irrigated from the same well; however, no data were available on the nitrate-nitrogen concentrations of the water over the past 12 years. The only alternative was to use data obtained during the study period and assume that it represented the earlier periods. Twelve water samples were collected from the Simas well during the study period and were found to have a mean nitrate-nitrogen concentration of 12.5 mg/l with a range of 11.4 to 14.7 mg/l. Using a mean concentration of 12.5 mg/l, the amount of nitrogen added in the irrigation water over 12 years was calculated at 1530 kg/ha for Block A and 1500 kg/ha for Blocks B and C. These calculations were based on a 30% runoff which was measured for this furrow irrigated field in another study. Total nitrogen inputs are given in Table 5.

The assumption also had to be made that nitrogen removed by crops and leaching characteristics of the field had not changed significantly over the past 12 years in order to calculate crop removal of nitrogen and nitrogen leached. The amount of nitrogen removed by a broccoli crop in Simas field in 1977 is given in Table 2. The yield was determined for nine study sites within the field. These sites (10 x 10 m plots) were the same as those used for soil sampling as discussed later. Broccoli samples were taken from each plot for nitrogen analyses. Nitrogen removals could then be calculated on the basis of yield and nitrogen content which average 5.2% for the field. The Simas data show that yield and nitrogen removals differ significantly between sites within the field. Thus, in conducting nitrogen studies one should recognize that nitrogen removals are not uniform within a management unit.

Table 2. Yield and nitrogen removal for nine sites in the Simas field.*

Site	Yield+	Nitrogen‡ Content	Nitrogen Removed
	T/ha	%	kg/ha
390	9.55 yz	4.88 z	49.86 z
391	11.03 xyz	5.18 yz	56.59 yz
392	11.42 xyz	5.17 yz	64.04 yz
393	8.45 z	5.69 y	49.23 z
394	12.06 xyz	5.28 yz	65.31 yz
395	12.57 xyz	5.12 yz	63.63 yz
396	14.97 x	5.05 yz	77.03 y
397	12.79 xy	5.32 yz	63.28 yz
398	11.00 xyz	4.91 z	56.26 yz

* Values followed by the same letter are not significantly different at the 1% level.

+ Determined from three replicates per plot.

‡ Determined from nine replicates per plot and expressed on a dry weight basis.

As no information was available on nitrogen removal by past crops at Simas, nitrogen removals were calculated using nitrogen contents for the same crops determined during studies at other locations. The nitrogen values used were 5.2% for broccoli, 3.6% for celery and lettuce and 4.8% for cauliflower (expressed on a dry weight basis). These calculations resulted in crop removal values of 2545 kg nitrogen/ha for Block A, 2380 kg nitrogen/ha for Block B and 2310 kg nitrogen/ha for Block C.

As mentioned previously the nitrogen leached was determined by soil sampling and analyses. Within the Simas field nine sites were sampled. Three holes were drilled at each site and soils were collected in 0.30-m increments from the 1.0-m depth to the water table, generally 4.0 m. These samples resulted in an estimate of the spatial variability of the nitrogen leaching within the field. The temporal variability was studied by collecting samples at three times during 1976-1977. A summary of the nitrate-nitrogen and chloride data for soil solutions below 1.0 m at Simas field is given in Table 3. The analyses of variance of the nitrate data show significant difference between sites and times. Mean nitrate-nitrogen concentrations for the nine sites ranged from 30.6 to 81.2 ppm for a field mean of 41.1 if one assumes that each site represents an equal area of the field.

Nitrate-nitrogen concentrations over the field as a whole decreased significantly for the third sampling period. A possible explanation for this was a reduced fertilizer application during the preceding crop. From Table 1 it can be seen that fertilizer nitrogen was applied at a rate of approximately 125 kg/ha for the 1976 lettuce crop which was about one-half or less of the rates used for previous crops. Chloride concentrations in the soil solutions were not significantly different over the same time period (7.3 vs 7.4 meq/l) which would rule out an increased leaching volume to dilute the nitrate-nitrogen.

The data given in Table 3 were used to calculate the nitrogen leached at each site in the Simas field (Table 4). The mean chloride concentrations in the soil solution below 1.0 m were used to calculate a leaching fraction at each site (Pratt et al, 1978) using the equation

$$\text{leaching fraction} = \frac{\text{chloride (irrigation-corrected)}}{\text{chloride (soil solution)}} .$$

Table 3. Analysis of variance for nitrate-nitrogen and chloride data for three sampling times for the Simas field. Values are mean concentrations in soil solution (ug/ml) below the 1.0-m depth.

	Site									Field as Whole
	390	391	392	393	394	395	396	397	398	
					Nitrate-nitrogen					
Significance+	N.S.	***	***	*	N.S.	**	***	*	**	***
Time 1	78.7z	58.7x	45.8y	46.7y	43.5z	44.3y	33.6y	36.1y	47.0y	48.3y
Time 2	89.0z	40.6y	54.2y	41.9yz	40.5z	40.5y	43.5y	34.9y	31.2z	46.2y
Time 3	76.2z	17.5z	20.5z	32.5z	33.2z	22.7z	17.6z	20.9z	19.1z	28.9z
Mean‡	81.2w	38.9xy	40.2x	40.4x	39.1xy	35.8xyz	31.6yz	30.6z	32.4xyz	
					Chloride					
Significance	***	***	*	***	**	*	***	N.S.	N.S.	N.S.
Time 1	7.1z	10.3y	6.2z	7.8z	7.1z	5.9z	5.9y	6.9z	8.1z	7.3z
Time 2	9.8y	6.8z	7.9y	9.3y	6.5z	7.2yz	3.8z	6.9z	7.9z	7.3z
Time 3	10.3y	5.8z	6.6z	7.2z	9.1y	7.7y	6.7x	7.0z	6.6z	7.4z
Mean	9.1w	7.6xy	6.9y	8.1x	7.6xy	6.9y	5.5z	6.9y	7.5xy	

+ Significance of differences between mean concentrations with time at each site and for the field as a whole are indicated here. Values within each site followed by the same letter are not significantly different at the level indicated.

‡ Site means followed by the same letter are not significantly different at the 5% level.

Again, data for the chloride concentrations of the irrigation water over the 12-year period were lacking. Therefore, the assumption was made that chloride concentrations in the irrigation water had not changed significantly over the 12 years. During 1976-77 twelve irrigation water samples were collected at Simas and analyzed for chloride. The concentration mean was found to be 4.8 meq/l with a range of 4.6 to 5.0 meq/l. This input chloride concentration was then corrected for crop removal using chloride contents of 0.66% for broccoli, 0.34% for cauliflower, 2.48% for lettuce and 3.20% for celery (dry weight basis).

Table 4. Nitrogen leached at each site in Simas field.

Site	Block	Chloride*	Chloride (irrigation-corrected)	Leaching Fraction	Nitrogen Leached
		----------meq/l----------			kg/ha
390	A	9.1	4.44	.49	4834
391	B	7.6	4.46	.59	2742
392	C	6.9	4.54	.66	3177
393	A	8.1	4.44	.55	2702
394	C	7.6	4.54	.60	2806
395	C	6.9	4.54	.66	2829
396	C	5.5	4.54	.83	3133
397	B	6.9	4.46	.65	2376
398	A	7.5	4.44	.59	2340

* In the soil solution below the root zone.

Leaching fractions for the different sampling sites ranged from .49 to .83 with a field mean of .62 based on the assumption that each site represented an equal area in the field. While at first this leaching fraction appears to be quite large, a companion study conducted at the Simas field to evaluate the irrigation system found deep percolation accounted for more than 50% of the applied water. The nitrogen leached was then calculated on a product of the leaching fraction, water applied and mean nitrate-nitrogen concentrations in the soil solution below 1.0 m (Table 4).

A summary of the measured and calculated nitrogen inputs and outputs at Simas field is given in Table 5 for each site and the field as a whole.

Table 5. Summary of nitrogen inputs and outputs at each site in the Simas field for 12 years.

	Inputs		Outputs*		
Site	Fertilizer	Water	Crop	Leached	Unaccounted-for
	kg/ha				
390	7052	1525	2545 (30)	4834 (56)	1198 (14)
391	6368	1500	2380 (30)	2742 (35)	2746 (35)
392	6459	1500	2310 (29)	3177 (40)	2472 (31)
393	7052	1525	2545 (30)	2702 (31)	3330 (39)
394	6459	1500	2310 (29)	2806 (35)	2843 (36)
395	6459	1500	2310 (29)	2829 (36)	2820 (35)
396	6459	1500	2310 (29)	3133 (39)	2516 (32)
397	6368	1500	2380 (30)	2376 (30)	3112 (40)
398	7052	1525	2545 (30)	2340 (27)	3692 (43)
Field	6636	1508	2404 (30)	2993 (37)	2747 (33)

* Values in ( ) are outputs as percent of nitrogen input.

For the field as a whole, crop removal amounted to 30% of the applied nitrogen and 37% of that applied was leached. The unaccounted-for nitrogen which in nitrogen balances is attributed to denitrification ranged from 14 to 44% of the applied nitrogen for a field average of 34%. When interpreting these figures one must keep in mind that all errors in measurement of other inputs and outputs are combined into the unaccounted-for category.

Direct field measurements of denitrification were made at site 396 during June to October, 1977 when a celery crop was growing in the field. For 335 kg nitrogen applied/ha, measured losses of nitrogen by denitrification amounted to 51 kg/ha or 15.2% of the applied nitrogen. Evidence was also obtained (Ryden et al, 1978b) that denitrification losses that occur during wet periods (winter rains) increase the proportion of applied nitrogen lost as a result of denitrification. At the same site in the Simas field, the denitrification loss between June, 1977 and February, 1978 was approximately 135 kg/ha or 29% of the nitrogen applied during that period. Other data obtained in these studies indicate that for similar irrigation management on similar soils, the amount of nitrogen lost by denitrification is fairly similar within quite a wide range of fertilizer inputs. Thus, if 50 kg were denitrified with an application rate of 200 kg (a rate more typical of lettuce and broccoli) the percent loss would be 25%. These observations suggest that the value of 34% obtained for unaccounted-for nitrogen in the nitrogen balance for the Simas field as a whole is not at all unreasonable.

The analyses of the data obtained for Simas field lead to the following conclusions:

(1) Crop removal of nitrogen varies within a management unit. Therefore, sampling programs must be designed to account for the variability when developing a nitrogen balance for the unit.

(2) Spatial and temporal variability of nitrate concentrations in soil solutions leaching below the root zone were observed and have implications relative to a steady-state model. By sampling at various points in time the relia-

bility of nitrate data obtained is increased. However, it must be kept in mind that this has implications relative to costs of the studies.

(3) A mean leaching fraction for the field obtained by chloride balances agreed with that obtained by an evaluation of the irrigation system efficiency. Within the field the leaching fractions were quite variable and were generally related to field position, i.e. greater at the head end of field.

(4) The field nitrogen balance attributed 30% of the nitrogen output to crop removal, 36% to leaching and 34% to denitrification. Direct field measurements of denitrification found that 29% of the applied nitrogen was lost at one site in the Simas field over an eight-month period.

BALANCE FOR THE VALLEY

Data obtained for the Simas field, data for seven other similar intensive study areas and data from some supplemental sampling have been combined with other existing data to develop a nitrogen balance for the entire drainage basin. As this balance is not yet complete only some of the data relative to the irrigated lands will be presented.

A breakdown of the areas in major crops in the valley for 1976 is given in Table 6. Broccoli, cauliflower and head lettuce account for approximately 80% of the area in vegetable crops. Seed beans, lima beans and irrigated pasture were the main field crops grown. Of the fruit crops, grapes made up most of the area.

The yields for the crops listed in Table 6 were taken from the Agricultural Commissioners' reports for Santa Barbara and San Luis Obispo Counties for 1976 and are averages for the entire basin. Average fertilizer nitrogen application rates as given in Table 6

Table 6. Cropping summary of major crops for Santa Maria Valley in 1976.

Crop	Hectares	Yield	Fertilizer nitrogen applied*	Nitrogen Removal by Crops
		metric ton/ha**	kg/ha	kg/ha
Vegetable Crops				
Artichokes	178	11.4	179	48
Broccoli	5,663	11.6	280	58
Cabbage	273	52.7	280	123
Carrots	475	40.8	134	78
Cauliflower	1,623	31.4	392	128
Celery	524	75.6	336	136
Lettuce, leaf	241	35.9	168	52
Lettuce, head	4,867	33.6	224	29
Peppers, dry	294	17.2	252	115
Potatoes	753	35.0	280	113
Spinach	184	20.2	224	91
Tomatoes	108	38.6	157	73
Field Crops				
Beans, seed	1,221	2.1	67	78
Beans, small, white	81	1.9	67	73
Beans, lima	1,666	2.4	67	91
Hay, alfalfa	202	15.7	0	453
Pasture, irri.	2,222	2.2	80	36
Sugar beets	405	64.5	280	109
Fruit Crops				
Lemons	81	24.2	224	12
Strawberries	299	47.1	336	52
Grapes	1,739	3.5	67	14

* Does not include nitrogen from irrigation water, rainfall, etc.

**Yields are based on material leaving field.

were developed from published data and information supplied by farmers and extension personnel. Many areas of the valley are used to produce more than one crop/yr. Thus, the data on fertilizer applied are not necessarily the amount applied/ha of land/yr. Some estimates indicate 1.3 crops/yr on the average for the valley.

During the 2 years of the Santa Maria project, plant samples of all major crops were collected and analyzed. These data were then used to calculate nitrogen removal from fields as given in Table 6. The average nitrogen removed by all crops was 31% of the fertilizer input. If the nitrogen applied in the irrigation water was included the crop removal value for the valley would be reduced to close to 24%. From the data in Table 6, the fertilizer nitrogen applied in 1976 was 4770 metric tons. Based on an average water application of 75 cm/ha, 80% application efficiency and a nitrate-nitrogen content of 10 mg/l, the nitrogen applied in the irrigation water was calculated as 1390 metric tons. Thus, the total nitrogen input to the irrigated lands of the valley was 6160 metric tons.

At the present time nitrogen losses from the irrigated lands in the Santa Maria Valley appear to breakdown as follows. From the data in Table 6, the crop removal of nitrogen was calculated as 1500 metric tons. Based on nitrate-nitrogen concentrations under a number of study areas in the valley and a leaching fraction of .25 as calculated from chloride concentrations in leaching solutions, the amount of nitrogen leaching below the root zones of irrigated lands in the valley was calculated as 2380 metric tons. The combined figures for crop removal and leaching was 3880 metric tons leaving 2280 metric tons in the unaccounted-for category. These resulting figures are 24% of the applied nitrogen removed in harvested crops, 39% leached below root zones and 37% unaccounted-for, which can be attributed to denitrification.

The first approximation of a nitrogen balance for the Santa Maria Valley has been presented here. The amounts of nitrogen attributed to the various sinks for the valley as a whole were not greatly different from those of the Simas field. This is not too surprising as the Simas Field is very typical of the vegetable crop production area and the crops grown in the rotation (broccoli, celery, lettuce, and cauliflower) at Simas account for some 55% of the crop production in the valley. Thus, these crops dominate any balance for the valley. Additional approximations for the nitrogen balance for the valley will be made as more data on individual study areas are obtained and will be reported in the final report of the NSF project "Nitrates in effluents from irrigated lands".

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ECONOMIC IMPACTS OF CONTROLLING NITROGEN CONCENTRATION AND OTHER WATER QUALITY DETERMINANTS IN THE YAKIMA RIVER BASIN

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ABSTRACT

The nonpoint source nature of water quality degradation caused by irrigation return flows makes control nearly impossible with traditionally effective measures, such as limitations or taxes on effluents. As a consequence, control or taxation of those inputs to production which are involved in effluent input-output relationships is a possible alternative. In the Yakima River Basin, three determinants of river water quality were identified: nitrogen concentration, river water temperature, and soil erosion by irrigation water. Among the pollution control policies considered for effectiveness, producer cost, and social cost were taxation of nitrogen fertilizer, increasing the charge for irrigation water, reduction of irrigation water rights, and restrictions on the types of irrigation systems used.

Results showed that policies which affect the level of one pollutant may or may not significantly affect the levels of others. Furthermore, policies taxing or charging for inputs, such as fertilizer and irrigation water, cause a substantial reduction in producer income if acceptable water quality is to be attained. Therefore, water quality policy formulation and evaluation should consider the interactions

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which exist among water quality determinants and considerations of both the expected benefits of improved water quality and the level and distribution of improvement costs.

INTRODUCTION

It has been traditionally assumed that the social benefits of irrigated agriculture far outweigh the social cost imposed by its contribution to environmental degradation. However, this assumption is now being questioned on many fronts. This paper relies on a study designed to estimate the agricultural contribution to water quality problems in the Yakima Basin of eastern Washington, to evaluate the efficiency and the impact on agriculture of alternative water quality improvement policies in the area, and to recommend policies which could improve water quality in this and similar irrigated river basins.

Water quality is a concept which transcends any single pollutant. In fact, many individual pollutants are interrelated chemically and biologically in the determination of water quality. Thus, policies designed to affect only one component of water quality may increase, decrease, or have no impact on other components of water quality. Three components of water quality were identified for evaluation in this study: river nitrogen concentration, river water temperature, and suspended solids resulting from soil erosion caused by irrigation.

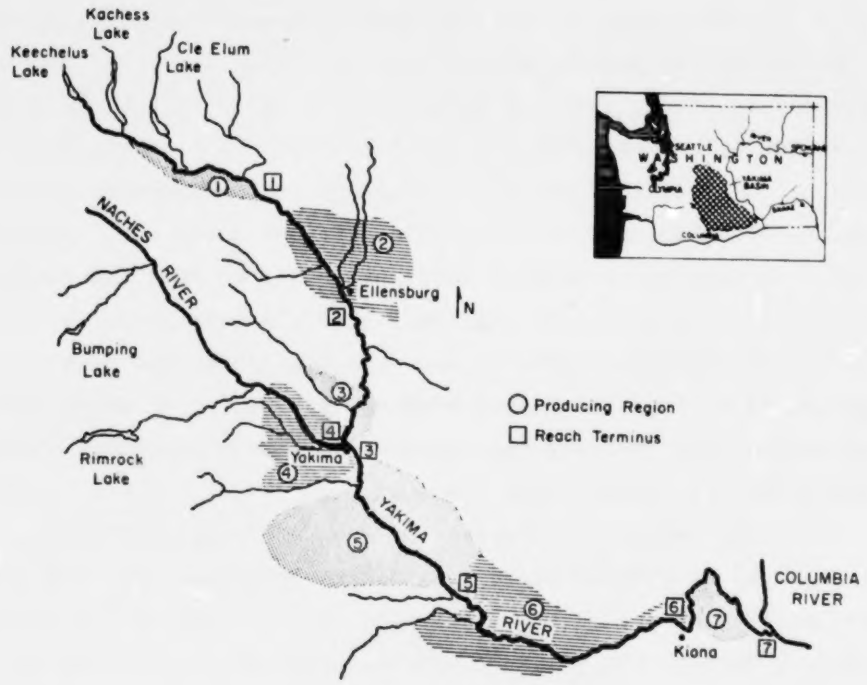
As a practical matter, irrigation related water quality degradation has been a difficult problem. The amendments to the Federal Water Pollution Control Act of 1972 (PL 92-500) are an important but perhaps unworkable attempt to address irrigation related pollution. The root of the problem is the nonpoint nature of such contaminants as nitrogen and sediment in return flow waters. As a result, it is difficult if not impossible to restrict, tax, or regulate such pollutants because the exact sources and extent of the losses cannot be identified. Hence, losses can only be controlled by secondary means; that is, by controlling input use or production practices related to the losses.

THE STUDY AREA

The Yakima River Basin is an intensively cultivated area of some 450,000 irrigated acres located in eastern Washington (Figure 1). Tree fruits and grapes comprise nearly 20% of the irrigated acreage; sugarbeets, hops, and mint comprise 15%; and potatoes and vegetables comprise 11%. The remaining 54% is divided among small grains, hay crops, and irrigated pasture. Abundant and inexpensive irrigation water has stimulated relatively inefficient water use. Approximately 5.3 acre feet of irrigation water are diverted from the Yakima River/irrigated acre (Copp and Higgins, 1974). A relatively small proportion of the canals and ditches in the area are lined, and most of the irrigation is by flood or furrow methods, with water diverted from the river in canals, and the part not consumed by plants returning through surface and subsurface return flow channels.

In the irrigation process, more water is applied than is consumptively used by growing plants. As a result, water not used passes through the soil below the root zone or runs off the land on its surface. Water passing through the soil picks up soluble compounds, some of which are nitrogen compounds applied as fertilizers. Runoff water may pick up some soluble compounds and may acquire suspended solids through erosion.

Because 80 to 90% of the water in the lower reaches of the Yakima River is irrigation return flow water during the late summer, the quality of the river water is a direct consequence of the quality of the return flow. Furthermore, the low summer flow volume in the river permits the water to warm substantially, diminishing its usefulness for recreation, fisheries, and promoting the growth of water-borne organisms. The specific determinants of water quality used in this


Kachess
Lake
Keechelus
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Cle Elum
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NACHES
RIVER
Bumping
Lake
Rimrock
Lake
Ellensburg
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Yakima
YAKIMA
RIVER
Kiona
COLUMBIA
RIVER
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1 2 3 4 5 6 7
○ Producing Region
☐ Reach Terminus
SEATTLE
WASHINGTON
OLYMPIA
YAKIMA
BASIN
SPOKANE
COLUMBIA

Figure 1. The Yakina River Basin.

study were August mean nitrate nitrogen concentration in the Yakima River, August mean maximum daily water temperature in the river, and annual soil losses from farmland. For the purpose of analysis, the river was divided into seven reaches with corresponding land areas as shown in Figure 1. Each land area (Regions 1 through 7) drains principally to its corresponding reach (Reaches 1 through 7).

ECONOMIC THEORY

Economic efficiency is generally maximized when pollution is directly controlled through taxation, restrictions, or "bribes" for abatement. However, agricultural effluents are not subject to the same types of constraints that are applicable to smokestack and sewage discharges because it is difficult, if not impossible, to identify the source of the discharges. Consequently, agricultural pollution abatement must be controlled through policies affecting the use of inputs causing the externality rather than policies directly affecting the externality.

Langham (1972) has shown that when pollution is a function of the use of one input, taxation or restriction on the use of that input is equivalent to controlling the pollution output itself from an efficiency standpoint. It can be shown that when pollution is a function of more than one input, the efficiency criterion can still be satisfied by appropriately taxing or restricting the use of all those inputs (Baumol and Oates, 1975; Pfeiffer, 1976). Income or cost distribution between the public and private sectors, however, depends on whether input use is controlled by taxation, restriction, or bribery. Also, some difficulty may be encountered in determining the appropriate set of taxes or restrictions on individual inputs which causes environmental standards to be met at minimum cost.

Further, while it is possible to determine the minimum cost means of achieving any given environmental quality standard, little light is shed on the appropriate standard to be set. While theoretical economics has approached this question, applied economics has not yet reached entirely satisfactory methods of application.

Finally, even if the physical and biological relationships and the benefits to society of differing levels of water quality are known, the optimal level of water quality and the impacts on agriculture which are required to attain it are predicted on predetermined prices for agricultural inputs and outputs. If the prices of factors of production or agricultural commodities change, either the optimal means of achieving a standard will change, the optimal standard itself will change, or both.

WATER QUALITY COMPONENTS

In the Yakima River, field losses of nitrate nitrogen[2] and eroded soil are two major factors affecting water quality. Both nitrogen and soil losses are relatively simple to measure in a laboratory situation, but quite difficult to predict under the variations of weather, cultural practices, and environmental conditions found in field situations.

Gossett and Whittlesey (1975) used data from Pratt and Adriano (1973) to estimate a nitrate nitrogen percolation regression equation for the Yakima Valley. The equation developed was:

$$N_L = .2\ (N_a Q_d)^{.712}$$

Where:

N_L = pounds of nitrogen leached/acre/year

N_a = pounds of nitrogen applied/acre/year

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[2] Nitrogen percolation losses are based on the weight of nitrogen lost, although it is assumed that all percolation losses are in the nitrate form.

Q_d = drainage volume in acres inches/acre/year

The major criticism of this function is that it is convex from above with respect to nitrogen application. To connect this intuitively illogical characteristic, the nitrogen loss estimation equation used in this study was:

$$N_L = .029\ (N_a)^{1.05}(Q_d)^{.7}$$

So, for example, on a crop to which 200 pounds of nitrogen is applied/acre/year and for which deep percolation is 12 acre inches/acre/year, estimated nitrogen loss is 51 pounds/acre/year.

Soil losses were determined using a procedure based on the Universal Soil Loss Equation developed by Wischmeier and Smith (1965). As suggested by Gossett and Whittlesey (1975) soil losses were estimated as linear functions of surface water runoff volume for each soil type and slope gradient in the Valley. Corrugated crops (alfalfa, pasture, orchard crops, etc.) were assumed to lose soil at a rate one-tenth as great as furrow crops (corn, vegetables, hops, etc.). Erosion from land irrigated by sprinkler or tailwater reuse systems was assumed to be negligible. Soil losses as high as 10 tons/acre were estimated on surface irrigated potatoes.

River water temperature was the third water quality parameter estimated. Briefly, changes in the temperature of a body of water are a function of the net change in stored energy in the water (from the sun), the volume and exposed surface area of the body, and the volume and temperature of inflowing water. Yakima River water temperature for August was estimated using equations developed by Raphael (1962).

ANALYTICAL CONSTRUCT

The analytical construct included a linear programming model of the agricultural sector of the Basin and the hydrology of the

Yakima River fortran-linked to a simulation model which determined values for the environmental quality parameters. While it is theoretically possible to include determination of environmental quality parameters within the linear programming model [see, for example, Kraft (1975)] using separable techniques, it was determined that such an inclusion would expand the model to a size too expensive to run repeatedly.

Within the seven agricultural regions in the Basin, constraints limiting irrigation water diversions, crop acreage and rotation, and irrigation system use were developed from historical data for each region. A solution to the linear programming model gave the profit maximizing combination of cropping activities, irrigation system use, and nitrogen fertilizer application subject to crop acreage, water rights, labor availability, and other constraints in each of the seven production regions within the Basin. Three irrigation systems--surface or rill, tailwater reuse, and sprinkler--and four levels of nitrogen fertilizer application were permitted for most crops. No nitrogen was applied on alfalfa. River water flow was estimated within the programming model at each reach terminus based on water usage in the upstream reaches.

Annual nitrogen lost to return flows in each region was estimated using the nonlinear exponential function discussed earlier. Soil losses were also estimated as noted. Data regarding annual and August water flows at the end of each reach, diversion and return flow volumes leaving and entering each reach, and annual sediment and nitrogen losses from each region were used as input to the water quality simulation model. Nitrogen concentration at each reach terminus was then estimated within the simulation model as a function of nitrogen entering the reach from the previous reach, nitrogen entering the reach with return flows, biological uptake of nitrogen, and river volume. Water quality as determined by sediment was estimated simply as annual sediment loss per irrigable acre.

August water temperature at each reach terminus was estimated as a function of river water temperature and volume entering the reach, reach length, river surface area, flow speed, August diversions, return flow volume and temperature, and net radiant energy.

POLICY OPTIONS

Policies for water pollution abatement were selected for their operational and administrative feasibility in the control of non-point source discharges and are designated Solutions 1 through 5.

1. Solution 1 represents existing agricultural production and environmental quality in the Yakima Basin.
2. Solution 2 imposed a tax on nitrogen fertilizer to control the level of its use for pollution abatement.
3. Solution 3 imposed a per acre foot charge for irrigation water delivered to the farm to reduce water use.
4. Solution 4 reduced water rights by a uniform percentage in all regions of the Basin.
5. Solution 5 combined a nitrogen fertilizer tax with a charge for irrigation water.

The stringency of each policy was increased incrementally until the specified level of environmental quality was reached. Specified environmental quality was a maximum August nitrogen concentration of 0.30 mg/1, maximum sediment loss of one ton/irrigable acre, and maximum water temperature of 70F. The nitrogen concentration goal is the level below which algae blooms will not occur (U.S. EPA, 1975). The water temperature goal satisfies the Washington State Department of Ecology requirement for a Class A stream. No standards exist for soil losses specifically, so a standard of approximately half the existing level was arbitrarily selected. Two measures of abatement cost were calculated: producer abatement cost and social abatement cost. Producer abatement cost was the reduction in farm income caused by a policy. Social abatement cost was the reduction of income to society calculated as producer abatement cost minus charges and taxes collected for abatement purposes.

Obviously, producer and social abatement costs are only measures of the direct impact on the agricultural sector or society of improved water quality. Secondary impacts, such as reduced employment, reduced gross business volume and personal income, and population changes could be estimated using an input-output technique and demographic models. Moreover, as noted earlier, the costs estimated in this study are based on a specific set of input and output prices. Changing these prices changes the results, but the general conclusions regarding policy effectiveness and relative abatement costs remain valid.

RESULTS

Solution 1, the benchmark or existing conditions solution is shown on each tabular summary of Solutions 2 through 5. Existing conditions result in a net crop income of $107 million on approximately 453 thousand acres in the Basin. Approximately 2.4 million acre feet of irrigation water are diverted each year, with a maximum August river nitrogen concentration of 0.87 mg/1, 1.74 tons of sediment lost/acre, and a maximum river temperature of 75.5F.

The nitrogen tax imposed in Solution 2 was believed likely to be effective particularly in the reduction of nitrogen concentration because nitrogen losses are in part a function of nitrogen fertilizer application. With a sufficiently high tax, it was believed likely that other environmental parameters would improve as a secondary impact of the reduction in irrigated acreage. A tax of $.60/pound resulted in the satisfaction of nitrogen concentration and water temperature goals as shown in Table 1. The sediment loss goal was not met even at a tax of $.70/pound because the adoption of improved

Table 1. Income, abatement cost, resource use and water quality determined with nitrogen taxes.

Item	Unit	Solution 1[a]	Solution 2 - Nitrogen tax ($/pound)						
			.10	.20	.30	.40	.50	.60	.70
Net crop income	$1,000	106,910	97,495	88,723	80,860	74,028	68,262	62,972	58,240
Irrigated acreage	1,000 acres	453	453	453	414	436	430	420	408
Producer abatement cost	$1,000	---	9,415	18,187	26,050	32,882	38,648	43,938	48,669
Social abatement cost	$1,000	---	70	1,615	3,770	8,651	10,311	13,164	18,827
N applied/ acre	pounds	209	206	183	180	139	132	122	105
Water diverted	1,000 acre feet	2,393	2,400	2,393	2,244	2,057	1,995	1,916	1,849
Mile 30 flow, August	1,000 acre feet	100	100	100	130	158	164	176	186
N concentration, August[b]	mg/l	0.87	0.86	0.76	0.56	0.38	0.35	0.28	0.22
Sediment lost/ irrigable acre	tons	1.74	1.72	1.68	1.62	1.60	1.61	1.21	1.26
Maximum temperature	F	75.5	75.4	75.4	74.0	70.9	70.4	69.8	69.3

[a] Benchmark solution.

[b] Maximum concentration.

irrigation systems is not induced by a nitrogen tax. The nitrogen concentration goal was met by a combination of reduced nitrogen losses caused by lower fertilizer application rates and increased river water flow resulting from reduced irrigated acreage. Reduction of irrigated pasture crops constituted most of the lost acreage. Water temperature reduction was accomplished by increased river flow as documented by flow data at river mile 30 (the main downstream gauging station) shown in Table 1. Satisfying nitrogen concentration and water temperature goals was relatively expensive in Solution 2: a 41% reduction of producer income and a social abatement cost of $13.2 million.

The level of social abatement cost determined in Solution 2 resulting from progressively improving nitrogen concentration, water temperature, and sediment losses are shown in Figures 2, 3, and 4, respectively. As expected, the curves are generally convex to the origin, suggesting that social abatement costs increase at an increasing rate as water quality is improved. Caution should be used in the interpretation of Figures 2, 3, and 4. The social abatement cost depicted in each is the cost when the total social abatement cost is attributed to a single component of water. Therefore, it is not legitimate to add the abatement cost of reaching a specific nitrogen concentration goal to the cost of attaining a particular water temperature goal as that involves double counting of costs. The fact that policies designed to affect primarily one water quality component also affect the others makes evaluating the cost of changing only one impossible.

Solution 3 shown in Table 2 imposed a charge for irrigation water. Nitrogen concentration reduction occurred for two reasons. First, nitrogen loss was reduced by improved irrigation systems which lower deep percolation drainage volume. Second, the increased river flow caused by higher water prices diluted the nitrogen entering the river with return flows. Sediment losses were reduced by

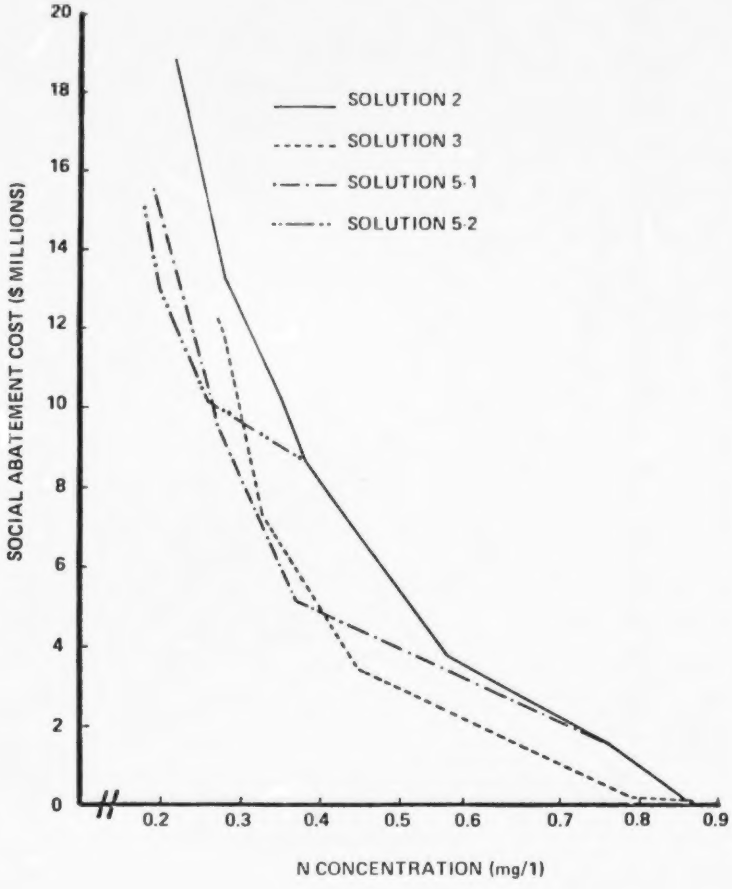

Figure 2. Social abatement costs for reducing nitrogen concentration.

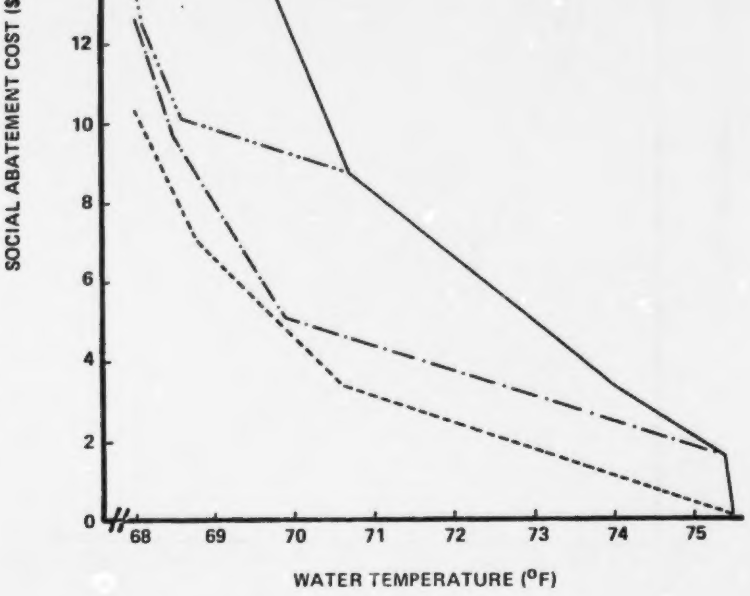

Figure 3. Social abatement costs for reducing water temperature.

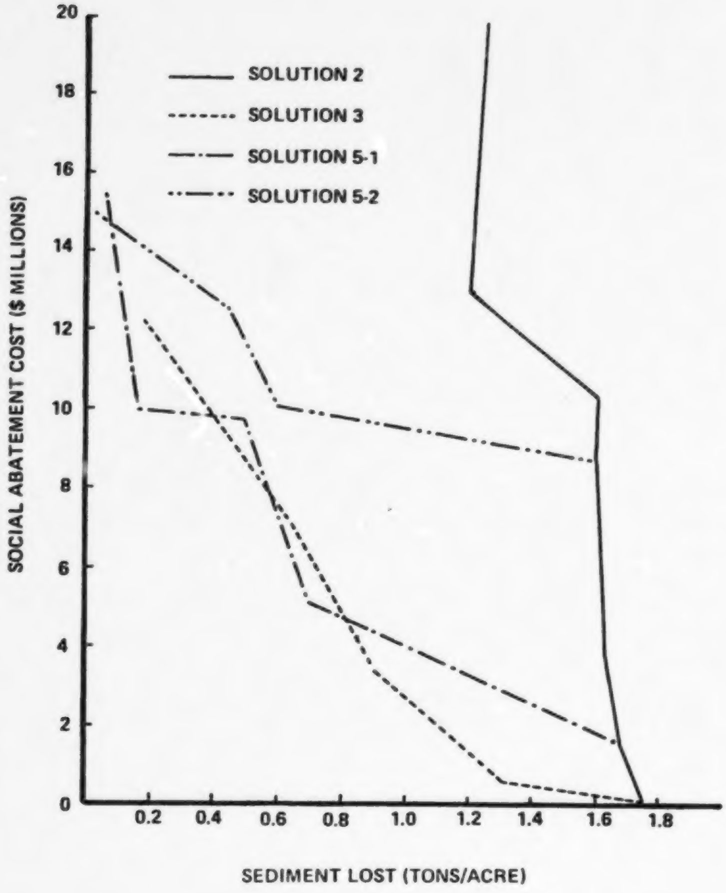

Figure 4. Social abatement costs for reducing sediment losses.

Table 2. Income, abatement cost, resource use and water quality determined with water charges.

Item	Unit	Solution 1[a]	Solution 3 - Water Charge ($/acre foot)			
			5	10	15	20
Net crop income	$1,000	106,910	98,043	90,064	83,963	78,959
Irrigated acreage	1,000 acres	453	453	395	319	261
Producer cost abatement	$1,000	---	8,867	16,845	22,946	27,951
Social abatement cost	$1,000	---	655	3,369	7,123	11,894
N applied/acre	pounds	209	206	213	226	269
Water diverted	1,000 acre feet	2,393	2,160	1,791	1,415	1,069
Mile 30 flow, August	1,000 acre feet	100	107	166	211	251
N concentration, August[b]	mg/1	0.87	0.79	0.45	0.33	0.28
Sediment lost/ irrigable acre	tons	1.74	1.31	0.88	0.64	0.20
Maximum temperature	F	75.5	74.7	70.6	68.8	67.6

[a] Benchmark solution.

[b] Maximum concentration.

the inducement of improved irrigation systems and practices. Water temperature was reduced because a water charge promotes water conservation and thus increases river water flow. A water charge of \$20/acre foot was sufficient to satisfy all three environmental goals, though the sediment goal was met with a \$10/acre foot charge and the temperature goal with a \$15/acre foot charge. Irrigated acreage, mostly irrigated pasture, was reduced by 43% with a \$20 charge while producer income was reduced 26%. Social abatement cost was \$11.9 million or 10% lower than Solution 2 with a \$.60 nitrogen tax of \$.60/pound, indicating that the water charge was a more efficient policy tool. Social abatement cost is shown graphically in Figures 2, 3, and 4.

Solution 4 reduced water rights by a uniform percentage in all regions of the Basin. Results of Solution 4 are summarized in Table 3. Water quality goals were met in Solution 4 for the same reasons they were met in Solution 3: reduced irrigation water use. A 50% reduction of water rights was required to satisfy all environmental goals, although the sediment loss goal was met by a 30% reduction and the temperature goal by a 40% reduction. The 50% reduction of water rights entailed a 36% reduction in irrigated acreage and a 16% reduction in farm income. The producer abatement cost was lower with a water rights reduction than with a water charge because restriction of rights does not involve the collection of taxes as a water use limiting mechanism. However, social abatement cost (the same as producer abatement cost in this case) was higher than either the fertilizer tax or the water charge. Uniformly reducing water rights in all regions caused an inefficient allocation of irrigation water among regions. Those regions where the marginal value of water was high, such as tree fruit regions, were required to reduce usage by the same proportion as those regions where its marginal value was relatively low. The inefficiency was reflected by a relatively high abatement cost.

Table 3. Income, abatement cost, resource use and water quality determined with water rights reductions.

Item	Unit	Solution 1[a]	Solution 4 - Water Rights Reduction (%)				
			10	20	30	40	50
Net crop income	$1,000	106,910	106,224	104,649	101,713	96,124	90,317
Irrigated acreage	1,000 acres	453	452	417	381	336	288
Producer abatement cost	$1,000	---	686	2,261	5,197	10,785	16,593
Social abatement cost	$1,000	---	686	2,261	5,197	10,785	16,593
N applied/acre	pounds	209	206	211	216	222	237
Water diverted	1,000 acre feet	2,393	2,195	1,951	1,706	1,464	1,219
Mile 30 flow, August	1,000 acre feet	100	111	140	167	197	228
N concentration, August[b]	mg/1	0.87	0.76	0.56	0.44	0.36	0.30
Sediment lost/ irrigable acre	tons	1.74	1.12	1.05	0.61	0.38	0.23
Maximum temperature	F	75.5	74.0	71.6	70.1	68.9	67.8

[a] Benchmark solution.

[b] Maximum concentration.

Solution 5 combined a nitrogen fertilizer tax with a charge for irrigation water for pollution abatement purposes. Results are shown in Table 4. Economic theory shows that limiting the use of all inputs functionally related to an externality can lead to a more efficient solution than that attained by limiting only a subset of those inputs. River nitrogen concentration is a function of both the use of nitrogen fertilizer and irrigation water, albeit indirect. Sediment loss and water temperature are primarily functions of irrigation water use. Thus, the inclusion of a nitrogen tax with a water limiting policy would theoretically lead to less efficient control of these parameters than a water limiting policy alone.

A nitrogen tax of $.20 in conjunction with a $10/acre foot water charge (.20-10) or a nitrogen tax of $.40 in conjunction with a $5/acre foot water charge (.40-5) were sufficient to meet the environmental standards. The .20-10 policy met standards at a lower producer and social abatement cost. Producer abatement cost was relatively high, a 30% reduction in farm income as a result of taxing both fertilizer and irrigation water use. Social abatement cost, however, was lowest of all policies considered, $9.7 million. Figures 2, 3, and 4 show social abatement costs for varying levels of environmental quality determined with Solution 5. Figure 3 shows that within the relevant ranges, taxing both nitrogen and water (Solutions 5-1 and 5-2) resulted in more efficient control of nitrogen concentration than taxing either alone. Solutions 5-1 and 5-2 represent Solution 5 with nitrogen taxes of $.20 and $.40/pound, respectively, and with water charges variable. Figures 3 and 4 show that the combination policies less efficiently control temperature and sediment loss than a water charge alone, but were more efficient than a nitrogen tax alone. These results are consistent with the functional relationships between inputs and environmental quality and the theoretical discussion presented earlier.

Table 4. Income, abatement cost, resource use and water quality determined with combination policies.

Item	Unit	Solution 1[a]	Solution 5 - Nitrogen tax ($/pound)--Water Charge ($/acre foot)						
			.20-5	.20-10	.20-15	.20-20	.40-5	.40-10	.40-15
Net crop income	$1,000	106,910	80,539	75,233	70,276	65,548	68,350	63,529	59,103
Irrigated acreage	1,000 acres	453	386	296	296	237	295	276	259
Producer abatement cost	$1,000	---	26,371	31,676	36,621	41,349	38,560	43,380	47,807
Social abatement cost	$1,000	---	5,061	9,664	9,921	15,654	10,149	12,535	15,021
N applied/ acre	pounds	206	190	203	203	238	198	197	195
Water diverted	1,000 acre feet	2,293	1,749	1,344	1,317	983	1,356	1,229	1,133
Mile 30 flow, August	1,000 acre feet	100	176	219	219	263	220	233	244
N concentration, August[b]	mg/l	0.87	0.37	0.27	0.27	0.21	0.26	0.22	0.20
Sediment lost/ irrigable acre	tons	1.74	0.69	0.50	0.16	0.06	0.60	0.44	0.03
Maximum temperature	F	75.5	69.9	69.5	68.5	67.4	68.5	68.1	67.8

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[a] Benchmark solution.

[b] Maximum concentration.

CONCLUSIONS

This research has shown that it is possible to substantially improve water quality in the Yakima River by controlling agricultural inputs or activities. The environmental improvement was accomplished with a reduction of farm income ranging from 16 to 41%, depending on the policy employed. In addition to reducing farm income, these policies would also impose a burden on the agricultural sector in the form of reduced land values, and decreased activity in agricultural input supply and agricultural processing firms. In all cases, the primary crops affected were low value forage crops. Consequently, the livestock sector would be affected most. The impact on agriculture would also be felt in nonfarm sectors of the economy because reductions of income in one sector indirectly reduce income in all sectors.

Net social cost ranged from $9.7 to $16.6 million depending on the policy used. A trade-off existed between economic efficiency and producer cost. For example, a combination nitrogen tax and water charge had the lowest net social cost but was relatively expensive to farmers. Reducing water rights uniformly had the highest social cost of policies evaluated but was least costly to farmers. This policy would minimize adverse agricultural impacts, but would result in less efficient organization of agricultural production.

The results show that considerable improvement in water quality, say 50% of the desired water quality standards, can be achieved with relatively minor costs to agriculture or the public. For example, from Figures 2, 3, and 4, it appears to be possible to attain 50% of the desired standards for a social cost of approximately $2.5 million. In contrast, complete attainment, or getting the next 50%, costs a total of $9.7 million or $7.2 million more. The fact that environmental improvement can be attained only at a cost is significant. More significant from a policy standpoint however, is the fact that costs increase at an increasing rate. Thus, while it may be relatively inexpensive to make substantial water quality improvements, reaching high levels of water quality may require sacrifices far out of proportion to the costs of initial improvements.

The policy which most efficiently reduced nitrogen concentration in the river was one which simultaneously imposed a tax on nitrogen fertilizer and a charge for irrigation water. This result is consistent with economic theory since the nitrogen concentration of the river is a function of both fertilizer application and water use. A water charge alone was less efficient than the combination policy, but more efficient than the other policies because nitrogen concentration is strongly affected by water use. When used exclusively, reducing water rights was more efficient than taxing nitrogen for low levels of abatement, but the nitrogen tax more efficiently met the 0.30 mg/l water quality standard.

Water temperature was largely a function of river flow volume, so policies that reduced irrigation diversions and thus water use reduced Yakima River water temperature most efficiently. A water charge was most efficient at temperatures below 71F because of the economic inefficiencies fostered by a uniform reduction of water rights. Policies that combined a water charge with a nitrogen tax approached the efficiency of policies dealing exclusively with water use when the water charge was sufficient to induce significant water conservation. The nitrogen tax policy less efficiently reduced water temperature since its major impact is on the use of nitrogen fertilizer, rather than the use of water.

Policy choices greatly affect reductions in agricultural income, and would surely affect the economy of the Basin differently. Policies involving taxes or charges for inputs more severely reduced producer income than policies which involved restrictions on input use. These distributional and secondary impacts may indeed be more important than strict economic efficiency in the final choice of the most appropriate policy for water quality improvement.

The choice of policies involves the evaluation of economic efficiency and income distribution. This analysis has generally shown that policies with a low social cost may have a high producer cost, while those with a low producer cost may have a higher social cost. For example, the uniform reduction of water rights had the lowest producer cost but the higher social cost for reducing nitrogen concentration. It is theoretically possible to design policies such as allocation of water rights where irrigation return flows are highest and such as monitoring and restricting the quality of return flow water leaving each farm. Such policies are both economically efficient and possess a low producer cost, but the transactions costs of administration often prohibit their use.

While several policies which could be implemented in the Basin have been analyzed, there exist other combinations and variations of these policies which might better combine the beneficial aspects of the policies while minimizing the detrimental aspects. For example, the improvement of irrigation efficiency alone was not sufficient to meet proposed environmental standards. No attempt was made to combine improved efficiency with other policies, though it is likely that the proposed standards could be met at a considerably lower cost by combining improved irrigation efficiency with input taxation or restriction. In fact, policies taxing or restricting water and nitrogen use may be more effective than indicated because no provision was made to include the improvements in water and fertilizer management that would likely occur except through the use of more efficient irrigation systems.

The social cost of the water rights reduction policy could be substantially lowered by reducing rights among regions in a fashion that would maintain equivalence of water value among regions. Such a procedure would involve greater than proportional decreases in water rights in parts of the Basin where the marginal product of water is low and smaller than proportional decreases of water rights in areas where the marginal product of water is high. This policy would approach the efficiency of a water charge without the adverse effects of a water charge on producer income. Distribution of income among regions would be less equitable than that caused by a uniform reduction of water rights, however.

This research demonstrates that the marginal value of water in the Yakima Basin approaches zero, probably because the marginal cost of water approaches zero in most cases. More importantly, the marginal value of water is low over a relatively large segment of its use. Nearly one-third of the irrigated land in the Basin produces pasture and grass crops returning a very low net income. Policy makers should remember that substantial improvements in water quality are possible by eliminating the low value uses of water with relatively minor reduction in crop sales and net income, though again, the distribution of reductions among individuals will not be uniform.

Economic theory prescribes that, in order to fully internalize a pollution problem, abatement activity should continue to the point that the marginal abatement cost equals the marginal benefit of the reduced external effect. This study estimated only direct social and producer costs of environmental improvement in the Yakima Basin. It is certain, however, that the costs estimated understate the true social and regional cost because secondary economic impacts were ignored. The Yakima Basin is a farm based economy, and the substantial reductions in agricultural activity would negatively affect farmland value, and the farm input supply sector, agricultural processing, and activities related to agriculture. These costs might be estimated by an input-output analysis. The benefits which would accrue to the public resulting from improved water quality in the Yakima River are less easily estimated, however. The state of economic science does not provide good measures of such benefits. In the case of the Yakima Basin, it is possible that benefits of environmental improvement would be relatively small because the river already meets the proposed standards above the City of Yakima.

Little has been said regarding the promise of water quality improvement through public investment in such facilities as water treatment plants or increasing upstream water storage facilities for water quality improvement purposes. The cost and effectiveness of irrigation return flow water treatment facilities has yet to be convincingly demonstrated on a large scale basis, so their use remains an engineering, rather than an economic problem. Some cost estimates have indicated that increased water flow to dilute existing effluent discharges might be feasible through enlargement of existing storage reservoirs. However, the propensity to utilize free (or governmentally subsidized) water resources for irrigation purposes and the likelihood that construction costs are underestimated cast doubts on the promise of such a solution.

Little also has been said here regarding the promise of other kinds of public or private investment in the form of research and development. There can be little question that technological developments can alter the inputs-outputs-effluents production relationship. A recent example is the development of soil nitrogen stabilizing compounds. The development of technologies which reduce externalities without the private and social sacrifices involved with reducing input use must be pursued.

Finally, examination of abatement costs shows that social cost is more a function of the acceptable level of environmental quality chosen than the method of abatement. While efficient abatement is desirable, other goals, such as producer income or agricultural output, may outweight the efficiency criterion, dictating both different levels of environmental quality and a less efficient method of pollution abatement.

RECOMMENDATIONS

Those unfamiliar with Pacific Northwest irrigated agriculture might find water delivery and application efficiency in the Yakima Basin and other areas appalling. Changes caused by the drought of 1976 demonstrate that crops in the Yakima Basin can be grown with substantial reductions in water diverted. Agencies involved in water management could encourage and promote irrigation district improvements, water conservation, and the development and use of more efficient and water conserving irrigation systems and practices. To the extent that such educational and promotional campaigns are effective, they should be pursued. However, in recognition of the fact that inputs with zero marginal cost will generally be used to provide their maximum total physical product, irrigation districts should be encouraged to charge customers on the basis of water delivered rather than acreage irrigated. The imposition of such a charge and the need for water flow gauging would increase awareness of water conservation from an economic and public relations standpoint. The improved water use efficiency that would surely result from even a nominal charge/acre foot of water would reduce losses of nitrogen and sediment, and increase river flow, allowing greater effluent dilution and lower water temperature.

It is also suggested that desired water quality standards be defined in terms of the gain in social welfare associated with irrigation return flow pollution abatement. The high cost to agriculture of meeting proposed standards makes the implementation of policies to meet such goals politically impossible and morally questionable unless concrete public benefits can be identified and at least partially evaluated in economic terms. While the deficiencies of this and related studies leave room for additional improvement in the determination of water pollution abatement costs, it is doubtful that further refinement of costs can further illuminate the desirability of environmental improvement in the absence of benefit estimation.

Finally, it is hoped that the more complete discussion and understanding of problems regarding agriculture and our environment, that this study has, albeit small, attempted to foster, will encourage the examination of issues from a variety of perspectives: economic, technological, social, and environmental. Only through the rational process of weighing alternatives can judgments be made for the betterment of mankind.

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